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Exhibit 10

NSARB-2022-003 (AQ#1444)

REPORT ON THE OUTCOMES OF CONSULTATION

AQ#1444

Submission to the Nova Scotia Aquaculture Review Board

Nova Scotia Department of Fisheries and Aquaculture

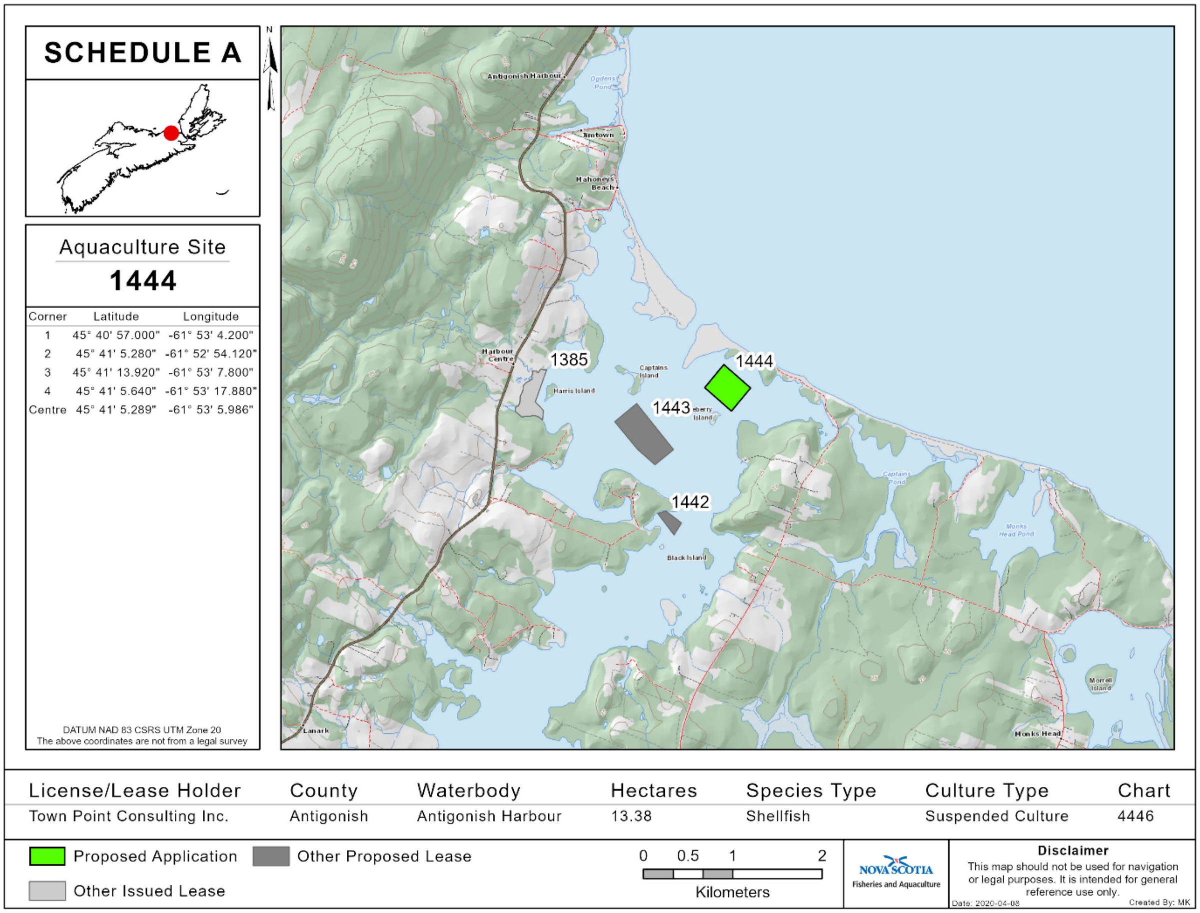
TABLE OF CONTENTS – CONSULTATIONS AQ#1444

LIST OF APPENDICES

1.0 APPLICATION DESCRIPTION

Ernie Porter of Town Point Consulting Inc. (TPCI) has applied for three marine aquaculture licenses and leases for the suspended cultivation of American oyster (*Crassostrea virginica*). The applications were received by the Department of Fisheries and Aquaculture (NSDFA) on January 27, 2020. All proposed sites are located in a body of water known as Antigonish Harbour, Antigonish County.

The differences between the site applications are the physical location, size of the sites and the production plan associated with each site. The following package consists of a report on the outcomes of consultations specific to AQ#1444.



Aquaculture Site
1444

Corner	Latitude	Longitude
1	45° 40' 57.000"	-61° 53' 4.200"
2	45° 41' 5.280"	-61° 52' 54.120"
3	45° 41' 13.920"	-61° 53' 7.800"
4	45° 41' 5.640"	-61° 53' 17.880"
Centre	45° 41' 5.289"	-61° 53' 5.986"

DATUM NAD 83 CSRS UTM Zone 20
The above coordinates are not from a legal survey

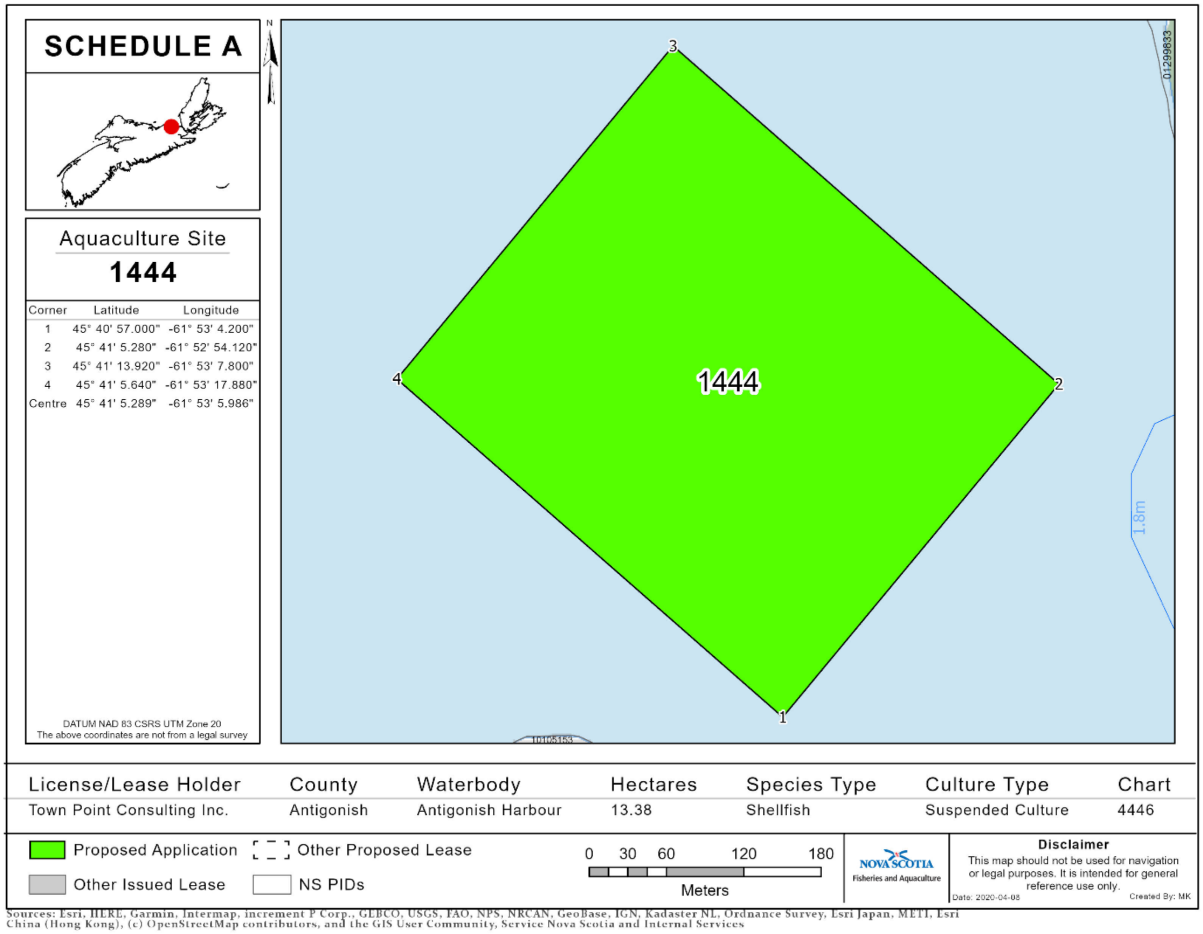
License/Lease Holder	County	Waterbody	Hectares	Species Type	Culture Type	Chart
Town Point Consulting Inc.	Antigonish	Antigonish Harbour	13.38	Shellfish	Suspended Culture	4446

🟩 Proposed Application
▨ Other Proposed Lease
▨ Other Issued Lease

Disclaimer
This map should not be used for navigation or legal purposes. It is intended for general reference use only.
Date: 2020-04-08
Created By: MK

Sources: Esri, HERE, Garmin, Intermap, increment P Corp., GEBCO, USGS, FAO, NPS, NRCAN, GeoBase, IGN, Kadaster NL, Ordnance Survey, Esri Japan, METI, Esri China (Hong Kong), (c) OpenStreetMap contributors, and the GIS User Community, Service Nova Scotia and Internal Services

Figure 1: Proposed lease AQ#1444. Please refer to NSDFAs Site Mapping Tool at https://novascotia.ca/fish/aquaculture/site-mapping-tool/ for an interactive map showing proposed leases.



Aquaculture Site

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Figure 2: Proposed lease AQ#1444. Please refer to NSDFAs Site Mapping Tool at https://novascotia.ca/fish/aquaculture/site-mapping-tool/ for an interactive map showing proposed leases.

2.0 CONSULTATION WITH MUNICIPAL, PROVINCIAL AND FEDERAL AGENCIES

NSDFA requested review of the application by Municipal, Provincial and Federal agencies listed in Table 1. An example of the "Network Memo" and "Network Agency Review of an Aquaculture Application" sent to the review agencies can be found in Appendix L.

These agencies provided advice based on their respective mandates to NSDFA on the proposed application. NSDFA worked with the applicant and the network agencies to respond to questions or comments regarding the application and to record any specific information relayed by the network agencies.

Table 1: List of Appendices

Fisheries and Oceans Canada	Appendix A
Canadian Food Inspection Agency	Appendix B
Transport Canada	Appendix C
Environment and Climate Change Canada – Canadian Shellfish Water Classification Program	Appendix D
Environment and Climate Change Canada – Canadian Wildlife Service	Appendix E
NS Department of Environment (Now Department of Environment and Climate Change)	Appendix F
NS Communities, Culture and Heritage (Now Department of Communities, Culture, Tourism and Heritage)	Appendix G
NS Department of Agriculture	Appendix H
NS Department of Lands and Forestry (Now Department of Natural Resources and Renewables)	Appendix I
Office of Aboriginal Affairs (Now Office of L'nu Affairs)	Appendix J
NS Municipal Affairs (Now Department of Municipal Affairs and Housing)	Appendix K
Sample of Network Memo and Network Agency Review of an Aquaculture Application	Appendix L

Summary of Network Consultations:

The following are summaries of the individual network agency consultations NSDFA undertook regarding the adjudicative boundary amendment application for lease #1444. Please see the appendices outlined in Table 1 to review the associated documents related to each of the following network agency summaries.

Fisheries and Oceans Canada (DFO) reviewed the application according to their legislative mandate which includes the *Fisheries Act*, *Species at Risk Act* (SARA), *Oceans Act* and applicable regulations. Initial questions and request for further information were submitted by DFO. These were forwarded to the applicant for response and the information was then provided to DFO.

DFO's assessment of the application resulted in their submission of information, advice, and recommendations which were provided to NSDFA for consideration and provided to the applicant as information awareness recommendations for the applicant to consider ensuring they would operate in compliance with DFO's legislated mandate. DFO Fish and Fish Habitat section identified no critical habitat or Species at Risk Act (SARA) listed species in the proposed lease area. If the application is approved, NSDFA will work with DFO to ensure that the advice and recommendations provided are appropriately incorporated into the required Farm Management Plan (FMP). **(See Appendix A)**

Canadian Food Inspection Agency (CFIA) submitted a network summary pertaining to AQ#1444 that indicates no concerns regarding the proposed development were identified. CFIA provided comments regarding the proponent's responsibility to culture shellfish in a manner that ensures they are safe for consumption and in compliance with the Canadian Shellfish Sanitation Program. **(See Appendix B)**

Transport Canada (TC) to date, has not provided a network response related to navigable waters concerns pertaining to the application. During a meeting with NSDFA, DFO and TC on May 9, 2019 TC had identified no issues related to the proposed sites and navigational issues. TC will complete their approval process once a decision is issued by the ARB. **(See Appendix C)**

Environment and Climate Change Canada – Shellfish Water Classification Program (SWCP) reviewed the application according to their legislative mandate and indicated that no concerns regarding the proposed development were identified and provided a comment that the shellfish harvesting classification for the area of the lease is currently Restricted. **(See Appendix D)**

Environment and Climate Change Canada - Canadian Wildlife Services Division (ECCC-CWS) Citing the Species at Risk Act (SARA), the Migratory Birds Conventions Act (MBCA) and provincial wildlife legislation, the network summary from CWS provides several operation recommendations for Town Point Consultants to follow in the event of the approval of their application. For lease AQ#1444, CWS has also recommended that a 300m buffer be in place for the areas of Gooseberry Island and Dunn's Beach based on information pertaining to presence of nesting gulls. The proposed lease AQ#1444 is within the recommended 300 meter buffer zone recommendations for these areas. (**See Appendix E and supporting videos)**

Nova Scotia Department of Environment Protected Areas Branch (NSDOE) (Now Department of Environment and Climate Change) reviewed the application according to their legislative mandate.

The initial NSDOE response was based on a broader scope than their regulations and jurisdiction dictate, including concerns relating to commercial and recreational fishing, and recreational boating. A subsequent network review was submitted by NSDOE staff that aligns with their regulatory authority and in which the revised NSDOE review identifies no concerns regarding the proposed development of Lease AQ#1444. **(See Appendix F)**

Nova Scotia Communities, Culture and Heritage (NSCCH) (Now Department of Communities, Culture, Tourism and Heritage) reviewed application for lease AQ#1444 according to their legislative mandate. CCH noted that they have concerns with the development of the lease and provided comments indicating that the area has elevated archaeological potential but given that the operation would utilize suspended culture the concerns were minimal. CCH provided recommendation that the lease operator should contact their Coordinator of Special Places, John Cormier if artifacts were encountered. **(See Appendix G)**

NS Department of Agriculture (NSDA) reviewed application for lease AQ#1444 application according to their legislative mandate and provided a network comments review that identifies no concerns from their department's perspective. The NSDA did note that agriculture in the surrounding area of Antigonish Harbour is active, and that this activity may increase in the future, and any noted that increased run-off from agricultural use upstream may have a negative effect on the water quality. **(See Appendix H)**

Nova Scotia Lands and Forestry (NSL&F) (Now Department of Natural Resources and Renewables) reviewed application for lease AQ#1444 according to their legislative mandate and identified concerns regarding the proposed development. Their concerns were identified under the Parks and Beaches Act, the Nova Scotia Wildlife Act, the Migratory Birds Convention Act, and the Species at Risk Act. NSL&F provide a list of requested information regarding these concerns to allow them to properly assess the impacts of the proposed development. NSDFA met with staff from NSL&F to determine the level of detail of the information that they were requesting and forwarded this request to the proponent to prepare a response. NSL&F found the proponents responses to the requested information acceptable and concluded that, with proper operational and managerial practices, that the development could proceed while taking the necessary recommendations to ensure that impacts to the ecosystem and wildlife were minimal and/or that the proper mitigation techniques were incorporated and practiced for all operations**. (See Appendix I)**

Nova Scotia Office of Aboriginal Affairs (Now Office of L'nu Affairs) reviewed the memo containing information relating to the application and provided advice on requirements for further consultation with the First Nations communities of Nova Scotia that might be impacted, or could provide feedback on the aquaculture lease development. **(See Appendix J)**

Nova Scotia Department of Municipal Affairs (Now Department of Municipal Affairs and Housing reviewed the memo of the proposed lease application and the detailed information pertaining to the lease. The memo serves as a notification of the proposed development to Municipal Affairs only. **(See Appendix K)**

3.0 SUMMARY OF CONSULTATIONS WITH THE MI'KMAQ OF NOVA SCOTIA

APPLICATION FILE NOs
AQ#1442, 1443, 1444

SUMMARY OF CONSULTATIONS WITH THE MI'KMAQ OF NOVA SCOTIA

Level of Consultation and the First Nations Communities Offered Consultation

The applications were sent to the Nova Scotia Office of L'nu Affairs (OLA) to screen the applications for Aboriginal consultation purposes. OLA found the applications to potentially involve impacts to Mi'kmaw Aboriginal and Treaty rights at the moderate end of the Haida spectrum.

The criteria used to assess the potential for intrusion on asserted or established Aboriginal or Treaty rights is further described in the initial offer to consult letter. These criteria included:

- The scope and scale of physical works required for the project;
- The proximity to Mi'kmaw communities;
- Regulatory requirements associated with the project (which estimate potential environmental impacts to waterways); and
- The potential for the existence of - and impacts to - heritage resources of Mi'kmaw origin within the project area.

On balance, NSDFA offered to consult the Mi'kmaq of Nova Scotia at a moderate level and reached out to Chiefs and Councils for reciprocity in the form of community-level and collectively held knowledge of potential adverse impacts to Aboriginal rights practiced within the project area which could be used to inform the results of our screening and open the consultation dialogue.

Consultation was initiated with the following groups:

- The 10 Chiefs and Councils of the Assembly of Nova Scotia Mi'kmaw Chiefs, including Membertou First Nation (under the August 31, 2010, Mi'kmaq-Nova Scotia-Canada Consultation Terms of Reference)
- Millbrook First Nation
- Sipekne'katik First Nation

Issues Raised by the Mi'kmaq of Nova Scotia During Consultation

The following issues were raised by KMKNO, the executive body that leads consultation efforts on behalf of the Assembly of Nova Scotia Mi'kmaw Chiefs:

1. Impacts to the Piping Plover species
2. Impacts to submerged Mi'kmaw archaeological resources.

NSDFA Assessment

Impacts to the Piping Plover Species

In response to concerns related to potential adverse impacts on Piping Plover in the project area, NSDFA provided the Mi'kmaq of Nova Scotia with the response received from Nova Scotia Department of Natural Resources and Renewables, provided during the standard network review process which did not anticipate any "undue negative effects on avifauna" resulting from the proposed aquacultural activities. The NSDFA also requested additional information from the Mi'kmaq of Nova Scotia to indicate how the proposed aquacultural activities set out in the applications could negatively affect Piping Plover and asked the Mi'kmaq to indicate how those negative effects could adversely impact the practice of Aboriginal and Treaty rights in the area of the proposed aquacultural activities. No further information was received by the Mi'kmaq of Nova Scotia. As such, the Department decided that this issue did not require further consultation or accommodation.

Impacts to submerged Mi'kmaw Archaeological Resources

In response to concerns about adverse impacts to potential submerged Mi'kmaw archaeological resources, NSDFA provided the Mi'kmaq of Nova Scotia with the response received from Nova Scotia Communities, Culture, Tourism and Heritage / CCTH (Previously known as Communities, Culture and Heritage / CCH) which was shared during the network review process. CCTH wrote:

"This is an area of elevated archaeological potential. Given this is a suspended culture set up, immediate archaeology concerns are minimal. However, please communicate to the license operator, that if at any time artifacts are encountered during operations, the Coordinator of Special Places, John Cormier should be contacted immediately."

The NSDFA requested that the Mi'kmaq of Nova Scotia further explain how the proposed aquacultural activities, which will employ suspended culture methods only, could impact Mi'kmaw Aboriginal and Treaty rights related to underwater archaeology in the area of the proposed aquacultural activities. The Mi'kmaq of Nova Scotia did not provide an explanation for how the specifically proposed suspended culture activities could impact rights related to underwater archaeology in the project area. The Mi'kmaq of Nova Scotia did however recommend a full Archaeological Resource Impact Assessment (ARIA) be undertaken.

Accommodation

The NSDFA decided to proceed with processing this application. In the absence of further information from the Mi'kmaq of Nova Scotia on how the proposed suspended culture activities associated with the applications could impact Mi'kmaw archaeological resources, the NSDFA has decided to follow the advice of the provincial regulator, CCTH, in requesting that the applicant contact the Coordinator of Special Places in the event any archaeological artifacts are encountered. The NSDFA considers this is an appropriate mitigation measure given the minimal potential impacts to underwater archaeology associated with suspended culture.

The 10 Chiefs and Councils of the Assembly of Nova Scotia Mi'kmaw Chiefs, KMKNO, Membertou First Nation, Sipekne'katik First Nation and Millbrook First Nation have been informed of this decision.

APPENDIX A: FISHERIES AND OCEANS CANADA

From: Greenwood, Megan N
Sent: October 7, 2020 2:26 PM
To: chris.mills@dfo-mpo.gc.ca; Braid, Leanna <Leanna.Braid@dfo-mpo.gc.ca>; Smith, Angela (CFIA/ACIA) <angela.smith@canada.ca>; david.macarthur@ec.gc.ca; Birch, Angela <Angela.Birch@novascotia.ca>; Kittilsen, Michael A <Michael.Kittilsen@novascotia.ca>; Cottreau-Robins, Catherine M <Catherine.Cottreau-Robins@novascotia.ca>; Blackburn, Lori M <Lori.Blackburn@novascotia.ca>; Boudreau, Louise O <Louise.Boudreau@novascotia.ca>; Gautreau, Rachel (EC) <rachel.gautreau@canada.ca>
Cc: Feindel, Nathaniel J <Nathaniel.Feindel@novascotia.ca>; Clancey, Lewis <Lewis.Clancey@novascotia.ca>; Ceschiutti, Robert <Robert.Ceschiutti@novascotia.ca>; King, Matthew S <Matthew.King@novascotia.ca>; Hancock, Bruce H <Bruce.Hancock@novascotia.ca>
Subject: AQ#1442, 1443, 1444 - New Application

Attention: Network Review Agencies

Please see the attached memo for three (3) Aquaculture Application Nos. 1442, 1443, 1444 in Antigonish Harbour, Antigonish County.

Please respond with your feedback by **December 7, 2020.**

The attached zipped shapefile requires specific geospatial software to open and is intended for convenient use by GIS professionals only.

Megan Greenwood

Licensing Coordinator
Nova Scotia Department of Fisheries and Aquaculture
1575 Lake Road
Shelburne, NS
B0T 1W0
Phone 902-875-7443
Fax 902-875-7429

Spencer, Amanda L
To: Clancey, Lewis
Subject: RE: Antigonish Harbour Aquaculture applications no. 1442, 1443 and 1444
From: Doucette, Renelle <Renelle.Doucette@dfo-mpo.gc.ca>
Sent: November 3, 2020 1:27 PM
To: Greenwood, Megan N <Megan.Greenwood@novascotia.ca>
Subject: Antigonish Harbour Aquaculture applications no. 1442, 1443 and 1444

Good afternoon Morgan,
I am assessing the new oyster aquaculture operation in Antigonish Harbour and have a few questions:
 If I understand correctly, site AQ1442 will be used as storage. How will it be done? Will material be stored on
the seafloor?
 In the application document, it states that the grow out sites (AQ1443, AQ1444) will be “floating from April to
October otherwise sunk”. What does this mean exactly? That the BOBR units will be spending the winter on the
seafloor?
If we issue a letter of advice, should we issue one letter for each site or put all three sites on the same letter?
Thanks in advance,

Renelle Doucette
Biologist | Biologiste
Fish and Fish Habitat Protection Program | Programme de la protection du poisson et de son habitat
Fisheries and Oceans Canada | Pêches et Océans Canada
343 ave. Université Avenue, Moncton NB E1C 5K4
Telephone | Téléphone 506. 851. 6914
renelle.doucette@dfo-mpo.gc.ca

From: Clancey, Lewis
To: Renelle.Doucette@dfo-mpo.gc.ca
Cc: Greenwood, Megan N
Subject: Re: Antigonish Harbour Aquaculture applications no. 1442, 1443 and 1444
Date: November 3, 2020 3:30:29 PM

Hi Renelle,

I’m responding to your questions to Megan Greenwood regarding the Antigonish leases 1442,1443, and 1444.

If I understand correctly, site AQ1442 will be used as storage. How will it be done? Will material be stored on the seafloor?

The applicant has indicated that AQ1442 will only be used to store oysters. Seed sized oysters and Market oysters will be submerged on the bottom and held in BOBR cages. During winter little, or no, growth will occur as oysters become dormant and do not feed during the winter months, during other times of the year the lease will provide a “warehousing” site for oysters harvested from the other grow-out leases as the await transfer to vehicles to move to a depuration site. The rationale for having a storage site is the ease of access during, and in preparation for, winter months. The close proximity provides a level of security in keeping an eye on the lease and provides easy access to marketable oysters from shore during harvest times. The lease is deep enough that seed and market oysters can be placed on bottom to protect them from ice damage. The applicant has indicated that no floating equipment will be onsite.

In the application document, it states that the grow out sites (AQ1443, AQ1444) will be “floating from April to October otherwise sunk”. What does this mean exactly? That the BOBR units will be spending the winter on the seafloor?

Yes, you are correct. Oysters stop feeding and growing once the seawater temperature falls below 5 degrees Celsius. In order to avoid damage from shifting ice it is standard practice for oyster farms that utilize floating cages/bags, or BOBR’s, to submerge them. The culture equipment is usually placed directly on bottom, however some farms suspend the cages/bags at the mid water column level as a predator control method.

If we issue a letter of advice, should we issue one letter for each site or put all three sites on the same letter?

Please issue one letter for each site.

Please feel free to contact me at anytime if you have further questions, or if I can be of assistance for this, or any lease application.

Thanks you,

Lew

Lewis Clancey
Aquaculture Development Advisor
NS Dept. of Fisheries and Aquaculture
1800 Argyle St.
WTCC 6th Floor, Suite 604,
Halifax, NS
B3J2R5
902 956 3839
lewis.clancey@novascotia.ca

From: Dwyer, Alan
To: Greenwood, Megan N
Cc: Doucette, Renelle
Subject: Town Point Inc.
Date: November 23, 2020 12:37:03 PM

Hello Megan,

Our Fisheries Protection Program have a couple of questions regarding the Town Point Consulting Inc. oyster lease applications in Antigonish Harbour. Hoping you could help us with the questions below?

Thanks,

Alan

- Has the proponent considered overwintering by lowering the cages approx. one foot from the seafloor instead of directly on the seafloor? Is this a possibility?

- Are the boundaries indicated in the Application dated September 4, 2020 final? i.e. they haven't changed since then?

Alan Dwyer

Area Manager, Resource Management, Gulf Nova Scotia / Chef intérimaire de la gestion des ressources, Secteur du Golfe Nouvelle-Écosse
Fisheries and Oceans / Pêches et Océans
Gulf Nova Scotia / Nouvelle Écosse – Golfe
2920, Highway 104 / 2920 route 104
Antigonish County, NS / Comté d'Antigonish (N.-É) B2G 2K6
Telephone / Téléphone: (902) 735-7114
Fax / Télécopieur: (902) 863-5818
E-mail / Courriel:Alan.Dwyer@dfo-mpo.gc.ca

Please call me if you wish to discuss this issue further. DFO Biologist, Dr. Barrell examined our growth units when he visited our proposed lease areas, he may be able add to the consideration of its reduced impact relative to prevailing growth units.

Best regards,
Ernie
Town Point Consulting Inc.
Ernie Porter, P.Eng.
President
Cellular: 902-471-3696
Email Address: ernieporter77@gmail.com

On Tue, Nov 24, 2020 at 2:11 PM Clancey, Lewis <Lewis.Clancey@novascotia.ca> wrote:

Hi Ernie,

DFO has asked for further information regarding the anchorage for overwintering lines, from the previous email questions.

The specifically ask to be supplied with information pertaining to, the total footprint of the blocks and anchors for each of the three sites for the Antigonish Harbour new aquaculture sites 1442, 1443, 1444

Could you provide a response for this information request?

Thanks,
Lew
Lewis Clancey
Aquaculture Development Advisor
NS Dept. of Fisheries and Aquaculture
1800 Argyle St.
WTCC 6th Floor, Suite 604,
Halifax, NS
B3J2R5
902 956 3839
lewis.clancey@novascotia.ca

From: Clancey, Lewis
To: Greenwood, Megan N
Subject: FW: Footprint - blocks and anchors
Date: November 24, 2020 2:13:09 PM

An info request for 1442, 1443, 1444 is below.

From: Doucette, Renelle <Renelle.Doucette@dfo-mpo.gc.ca>
Sent: November 24, 2020 9:30 AM
To: Clancey, Lewis <Lewis.Clancey@novascotia.ca>
Subject: Footprint - blocks and anchors

Good morning Lew,

Could I have the total footprint of the blocks and anchors for each of the three sites for the Antigonish Harbour new aquaculture sites 1442, 1443, 1444?

Thanks in advance,

Renelle Doucette
Biologist | Biologiste

Fish and Fish Habitat Protection Program | Programme de la protection du poisson et de son habitat
Fisheries and Oceans Canada | Pêches et Océans Canada

343 ave. Université Avenue, Moncton NB E1C 5K4
Telephone | Téléphone 506. 851. 6914
renelle.doucette@dfo-mpo.gc.ca

From:	Clancey, Lewis
To:	Doucette, Renelle
Cc:	Greenwood, Megan N
Subject:	RE: Footprint - blocks and anchors
Date:	November 25, 2020 8:10:52 AM

Hi Renelle,

Below is the answer that the applicant has provided for your question regarding blocks and anchors for AQ #s 1442, 1443 and 1444.
Lew

“Hi Lew

The following outlines our intended procedure for sinking growth units. I trust this explanation provides the requested information.

Our growth units are cylindrical cages 14"x42" which float at the surface by means of torpedo floats contained within the cage. The number of floats can be adjusted to suit the desired outcome such as more floats for high stocking density or heavier, more mature stock, fewer floats for seed and to achieve near neutral buoyancy when sinking.

In late fall, we plan to adjust floatation to enable easy sinking. Small concrete weights will be added to the static lines every few meters, the spacing depends on water depth.
We expect a spacing of about 3m and a weight size of about 12 square inches (concrete filled yogurt container or ice cream container). So, where water depth permits sinking only one of the static lines instead of both with a weight spacing of 3m, line length of 100m and weight area of 12 si would result in a bottom coverage of about 3sf per run. If both static lines required weights the bottom coverage would then be about 6 sf per run.

Our site #1 (AQ#1442), next to our nursery is intended for seed storage and over winter storage of marketable stock. It is not intended to be a grow out area, so we expect only about 30 runs to be sunk here which would result in bottom coverage of about 83sf or 8sm if single line sinking works or 16sm if both lines must be weighted.

Our site #2 (AQ#1443**),** Captains Island is planned to have 141 runs which would result in bottom coverage of 423 sf or 39sm if single line sinking works or 78sm if both lines must be weighted.

Our site #3, (AQ#1444) Gooseberry Island is planned to have 90 runs which would result in bottom coverage of 250 sf or 23sm if single line sinking works or 46 sm if both lines must be weighted.

If the equivalent oyster stock was grown in 6 bag Oyster Gro cages instead of our BOBR cages the over winter bottom coverage would be about 50 sf per run vs 3 sf assuming all Oyster Gro cages were positioned floats down. Our system reduces bottom impact by a factor of 16...one of its many benefits.

Please call me if you wish to discuss this issue further. DFO Biologist, Dr. Barrell examined our growth units when he visited our proposed lease areas, he may be able add to the consideration of its reduced impact relative to prevailing growth units.

Best regards,
Ernie”

From: Dwyer, Alan
To: Clancey, Lewis; Doucette, Renelle
Cc: Greenwood, Megan N
Subject: RE: DFO Fisheries Protection Program questions regarding your lease applications, Antigonish.
Date: November 23, 2020 2:30:50 PM

Thanks Lew and Megan,

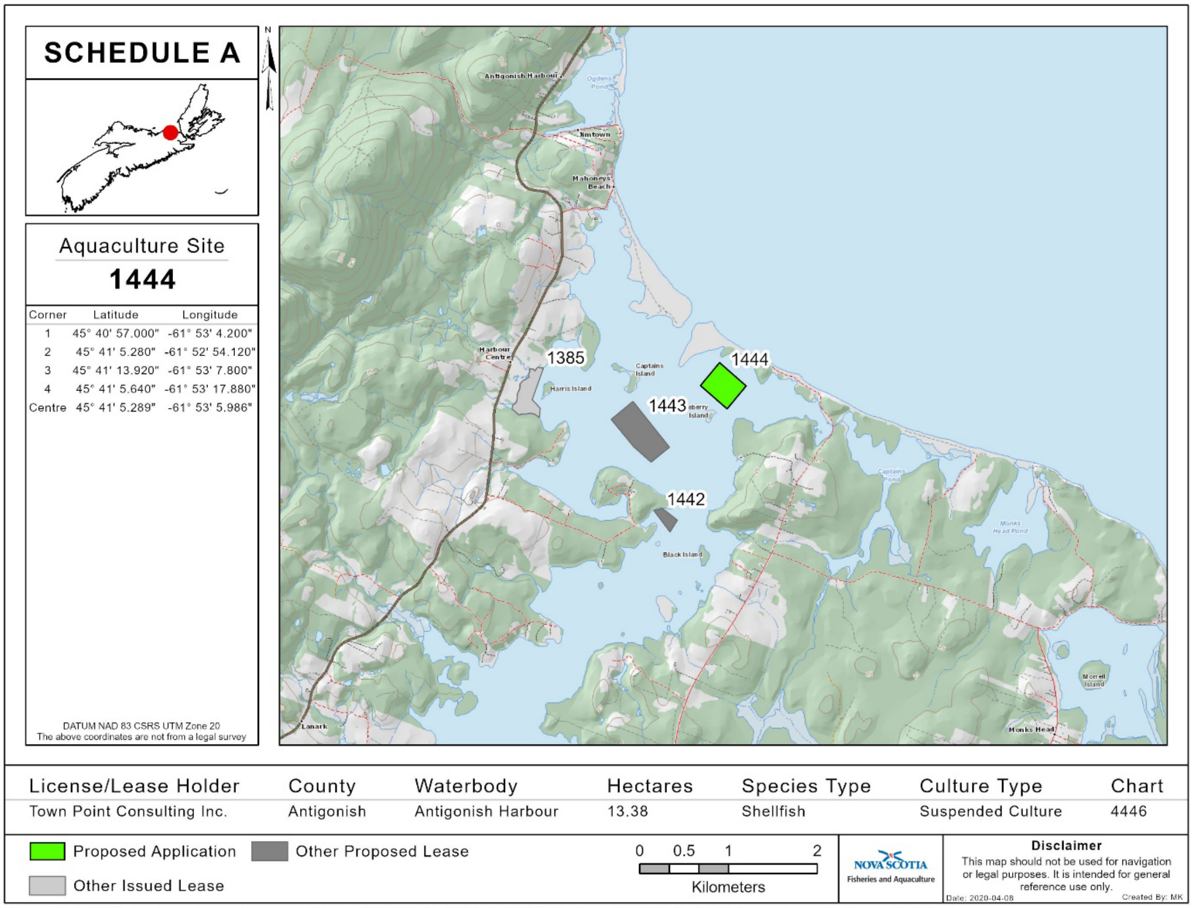
I assume that the boundaries never changed either ?

Alan

From:	Clancey, Lewis
To:	Dwyer, Alan; Doucette, Renelle
Cc:	Greenwood, Megan N
Subject:	RE: DFO Fisheries Protection Program questions regarding your lease applications, Antigonish.
Date:	November 23, 2020 2:33:02 PM
Attachments:	Schedule_A_1444.pdf Schedule_A_1443.pdf Schedule_A_1442.pdf

Hi Alan and Renelle,

I've attached the records we have on file for the leases. I believe these are up to date. Lew



SCHEDULE A

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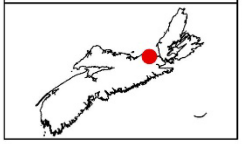
SCHEDULE A



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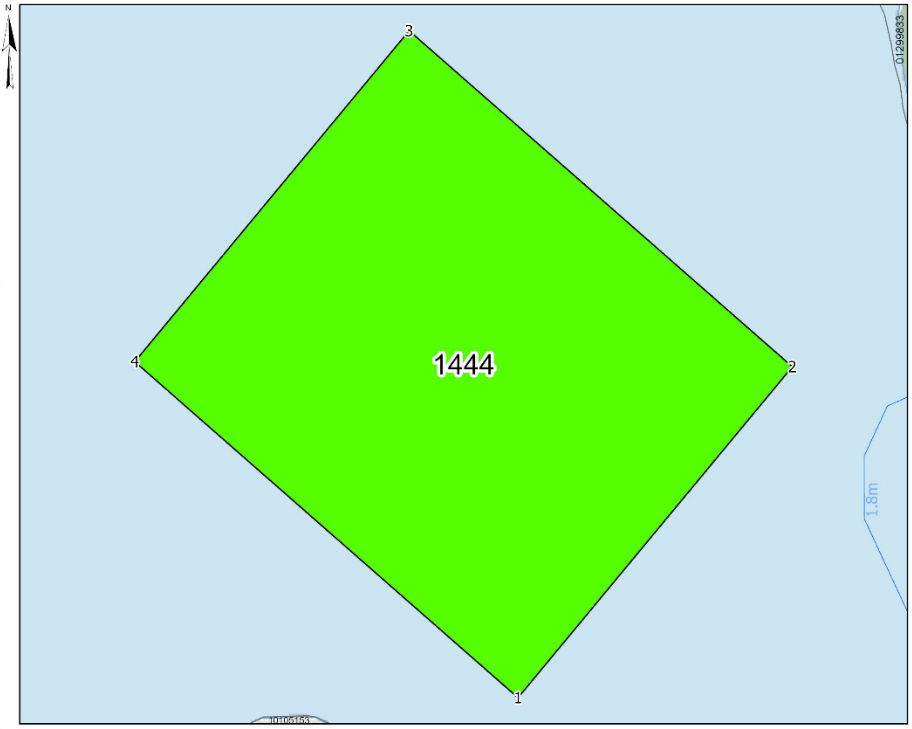
1	45° 40' 57.000"	-61° 53' 4.200"
2	45° 41' 5.280"	-61° 52' 54.120"
3	45° 41' 13.920"	-61° 53' 7.800"
4	45° 41' 5.640"	-61° 53' 17.880"
Centre	45° 41' 5.289"	-61° 53' 5.986"

DATUM NAD 83 CSRS UTM Zone 20
The above coordinates are not from a legal survey



License/Lease Holder	County	Waterbody	Hectares	Species Type	Culture Type	Chart
Town Point Consulting Inc.	Antigonish	Antigonish Harbour	13.38	Shellfish	Suspended Culture	4446

🟩 Proposed Application ⌜ ⌟ Other Proposed Lease

▨ Other Issued Lease ☐ NS PIDs

0 30 60 120 180
Meters

NOVA SCOTIA
Fisheries and Aquaculture

Disclaimer
This map should not be used for navigation or legal purposes. It is intended for general reference use only.

Date: 2020-04-08 Created By: MK

Sources: Esri, HERE, Garmin, Intermap, increment P Corp., GEBCO, USGS, FAO, NPS, NRCAN, GeoBase, IGN, Kadaster NL, Ordnance Survey, Esri Japan, METI, Esri China (Hong Kong), (c) OpenStreetMap contributors, and the GIS User Community, Service Nova Scotia and Internal Services

From: Doucette, Renelle <Renelle.Doucette@dfo-mpo.gc.ca>
Sent: November 25, 2020 10:17 AM
To: St.Louis, Danielle <Danielle.StLouis@novascotia.ca>
Subject: baseline monitoirng videos - Antigonish

Hi Danielle,
Do you know if the applicant has conducted a baseline monitoring video at a reference station 100m to 300m outside of each proposed lease site (#1442, 1443 and 1444)?
Or only within the sites?

Thanks,
Renelle Doucette
Biologist | Biologiste

Fish and Fish Habitat Protection Program | Programme de la protection du poisson et de son
habitat Fisheries and Oceans Canada | Pêches et Océans Canada
343 ave. Université Avenue, Moncton NB E1C
5K4 Telephone | Téléphone 506. 851. 6914
renelle.doucette@dfo-mpo.gc.ca

From: St.Louis, Danielle <Danielle.StLouis@novascotia.ca>
Sent: Wednesday, November 25, 2020 10:31 AM
To: Doucette, Renelle <Renelle.Doucette@dfo-mpo.gc.ca>
Cc: Clancey, Lewis <Lewis.Clancey@novascotia.ca>; Greenwood, Megan N <Megan.Greenwood@novascotia.ca>
Subject: RE: baseline monitoirng videos - Antigonish Hi

Renelle,

At the time, our baseline requirements for shellfish were not finalized, and therefore, a reference station was not required. If needed, we could request more baseline video to be completed.

Thanks,

Danielle

From: Doucette, Renelle
To: St.Louis, Danielle
Cc: Clancey, Lewis; Greenwood, Megan N
Subject: RE: baseline monitoirng videos - Antigonish
Date: November 25, 2020 11:12:10 AM

Ok, I understand. I was only curious.

Thanks for the quick reply,
Renelle

From: Dwyer, Alan <Alan.Dwyer@dfo-mpo.gc.ca>
Sent: December 3, 2020 2:31 PM
To: Greenwood, Megan N <Megan.Greenwood@novascotia.ca>; Clancey, Lewis <Lewis.Clancey@novascotia.ca>
Cc: Mills, Chris D <Chris.Mills@dfo-mpo.gc.ca>
Subject: TPCI Applications

Hello Megan and Lew,

We are still finalizing our comments and need a few more days to run them through the approval process. We are requesting an extension to provide our comments from Dec 7 to Dec 11.

We apologize for the delay and hope this request doesn't cause significant problems.
Regards,
Alan

Alan Dwyer

Area Manager, Resource Management, Gulf Nova Scotia / Chef intérimaire de la gestion des ressources, Secteur du Golfe Nouvelle-Écosse

Fisheries and Oceans / Pêches et Océans
Gulf Nova Scotia / Nouvelle Écosse – Golfe
2920, Highway 104 / 2920 route 104

Antigonish County, NS / Comté d'Antigonish (N.-É) B2G 2K6

Telephone / Téléphone: (902) 735-7114

Fax / Télécopieur: (902) 863-5818

E-mail / Courriel: Alan.Dwyer@dfo-mpo.gc.ca

From: Ceschiutti, Robert
To: Dwyer, Alan
Cc: Greenwood, Megan N; Clancey, Lewis; Chris Mills
Subject: FW: TPCI Applications
Date: December 3, 2020 3:04:26 PM

Hi Alan, thanks for the advance notice. Yes, we can extend the due date to December 11,

2020. Regards,
Robert Ceschiutti
Manager, Licensing and Leasing
NS Department of Fisheries and Aquaculture
1575 Lake Road
Shelburne, Nova Scotia
B0T 1W0
Phone: 902-875-7430
Cell: 902-874-0996
Robert.Ceschiutti@novascotia.ca

From: Clancey, Lewis <Lewis.Clancey@novascotia.ca>
Sent: December 3, 2020 2:35 PM
To: Ceschiutti, Robert <Robert.Ceschiutti@novascotia.ca>
Subject: FW: TPCI Applications

Network Agency Review of an Aquaculture Application

Agency	**Fisheries and Oceans Canada**
Division (if applicable)	
Date	**December 10, 2020**
File No.	**AQ# 1444**
Type of application	**New Marine Shellfish**
Information Provided	**Accompanying document**

Please provide comments, concerns, recommendations, or requirements on the above stated application for a marine aquaculture licence. Please include the criterion /criteria within your jurisdiction or mandate that your feedback is based upon. Similarly, if additional information is required to make a determination, please include the criterion /criteria within your jurisdiction or mandate that your request is based upon.

☐ No concerns regarding the proposed development
☐ Concerns with development are expressed below
☐ Request modifications to the proposed development (described below)
☒ Required or recommended conditions (accompanying document)
☐ Request additional information (described below)
☐ Request meeting with applicant and NSDFA (described below)
☐ No comments on the application

Comments, concerns, recommendations, and/or required conditions including the criterion /criteria within your jurisdiction or mandate that your feedback is based upon. (Attach comments if preferred, or add additional pages, as required.):

Fisheries and Oceans Canada have reviewed application AQ # 1444 in regard to potential conflicts with recreational, commercial, and communal commercial fishery activities.

There will be some conflict with recreational fishers who use boats to fish for striped bass and trout in Antigonish Harbour.

There are four oyster relay licences issued for Antigonish Harbour. Each licence holder has an additional harvester approved to fish under their respective licence for a total of eight in Antigonish Harbour. These licence holders may be displaced from their regular fishing areas and should be consulted.

There are other fishery species in the harbour including eel, smelt, gaspereau, and lobster where the amount of fishing activity fluctuates due to stock abundance and price. There is a potential for fisher displacement and gear conflicts on the fishing grounds.

While DFO recognizes Transport Canada's authority over safe navigation, there are commercial fishers who access wharves in the area that have provided DFO with an unsolicited email expressing concerns with navigation (attached).

The application has also been reviewed for potential impacts on fish and fish habitat including species listed under the Species at Risk Act.

Please be aware that the *Aquaculture Activity Regulations* (AAR) under the *Fisheries Act*, apply to the operation of a licenced aquaculture lease and constitute the authorization and set the conditions for compliance with Sections 35 and 36 of the Fisheries Act.

Considering the scope of the AAR's and the activities authorized under them, please find, accompanying this form, a document that describes a number of recommendations for the aquaculturist to follow to mitigate risks associated with potential effects to fish and fish habitat.

If you have any questions please contact me at 902-735-7114 or Alan.Dwyer@dfo-mpo.gc.ca .

Public Notice and Disclosure

As part of the process for deciding on an application, it may be necessary for the Nova Scotia Department of Fisheries and Aquaculture ("Fisheries and Aquaculture") to disclose the collected network review information to the applicant and other government bodies, including, if applicable, the Nova Scotia Aquaculture Review Board for use at an adjudicative hearing relating to the application in question.

In accordance with departmental policy, which seeks to promote public involvement in the process for deciding on aquaculture applications, Fisheries and Aquaculture will disclose aquaculture application information, including network review information, on the departmental website.

Privacy Statement

The network review information collected as part of an aquaculture application will only be used or disclosed by Fisheries and Aquaculture for the purpose of deciding on the application.

All application information collected is subject to the Freedom of Information and Protection of Privacy Act ("FOIPOP") and will only be used or disclosed in accordance with FOIPOP.

Fisheries and Oceans Canada
Pêches et Océans Canada

Fish and Fish Habitat Protection Program
343 Université Avenue
P.O. Box 5030
Moncton, New Brunswick
E1C 9B6

December 10, 2020

Your file
AQ#1444

Our file
20-HGLF-00438

Alan Dwyer
Area Manager, Resource Management, Gulf Nova Scotia
Fisheries and Oceans Canada, Gulf Nova Scotia
2920, Highway 104
Antigonish County, NS
B2G 2K6

Subject: Antigonish Harbour – Antigonish County – New Marine Aquaculture of American Oyster – Implementation of Measures to Avoid and Mitigate the Potential for Prohibited Effects to Fish and Fish Habitat

Dear Mr. Dwyer:

The Fish and Fish Habitat Protection Program (the Program) of Fisheries and Oceans Canada (DFO) received the proposal on October 13, 2020. We understand that the proponent (Ernie Porter, Town Point Consulting Inc.) proposes to:

- Install, operate and maintain a new aquaculture grow-out facility in Graham's Cove, Antigonish Harbour (45.684519 N -61.884644 W);
- Cultivate American oysters within the lease boundary in vexar mesh bags inside suspended cylindrical growth units measuring 36 cm diameter by 106 cm in length;
  - Lease #1444 would accommodate up to 90 lines at spacing of 10 m for a total of up to 9000 cages;
- Use helical anchors to secure the lines and buoys.

In addition, the following aquatic species are subject to the *Aquatic Invasive Species Regulations* and may be found in the vicinity of your proposed work, undertaking, or activity:

- Green crab (Carcinus maenas)
- Vase tunicate (Ciona intestinalis)
- Golden star tunicate (Botryllus schlosseri)
- Violet tunicate (Botrylloides violaceus)

Our review considered the following information:

- The commercial aquaculture licence/lease application documents for the suspended cultivation of American oyster in Grahams Cove, Antigonish County received on October 13, 2020;
- The Nova Scotia Fisheries and Aquaculture Memorandum to Aquaculture Network Agencies dated October 7, 2020, and received by the Program on October 13, 2020;
- The Science Advisory Report - Pathway of Effects for Finfish and Shellfish(CSAS) 2009/071;
- The Replacement Class Screening Report for Water Column Oyster Aquaculture in New Brunswick. Report of the Canadian Environmental Assessment Agency. Moncton, N.B. 2007;
- The Environmental Management Framework (EMP) and Standard Operation Procedures (SOP's) documents prepared June 2020;
- Baseline monitoring videos received from the Nova Scotia Department of Fisheries and Aquaculture on November 18, 2020;
- Additional information received by multiple emails, Teams chats and messages and telephone conversations between November 3 and December 3, 2020.

The proposal has been reviewed to determine whether it is likely to result in:

- The death of fish by means other than fishing and the harmful alteration, disruption or destruction of fish habitat which are prohibited under subsections 34.4(1) and 35(1) of the Fisheries Act; and
- Effects to listed aquatic species at risk, any part of their critical habitat or the residences of their individuals in a manner which is prohibited under sections 32, 33 and subsection 58(1) of the Species at Risk Act; and
- The importation, possession, transportation or release of aquatic invasive species set out in Part 2 of the schedule of the Aquatic Invasive Species Regulations which are prohibited under sections 6, 7, 8 and 9 of the Regulations; and
- The introduction of aquatic species into regions or bodies of water frequented by fish where they are not indigenous, which is prohibited under section 10 of the Aquatic Invasive Species Regulations.

The aforementioned outcomes are prohibited unless authorized under their respective legislation and regulations.

To avoid and mitigate the potential for prohibited effects to fish and fish habitat (as listed above), we recommend implementing the measures listed below:

- Place lines and configure units in such a way that long lines run parallel to tidal currents and prevailing winds to avoid overlap with the same area of macrophytes (e.g. eelgrass) overtime;
- Space rows of water column aquaculture work at a minimum of 3 m apart and not to exceed 50% of coverage of the surface area of the lease;
- Locate off-bottom aquaculture structures in areas with minimal eelgrass cover;
- Respect environmental and social standards as demonstrated by the Best Aquaculture Practises Certification Standards, Guidelines for growing sites in Nova Scotia.
- Minimize sedimentation of the waterbody during all phases of the installation, operation and maintenance of the aquaculture facility:

  - Regularly monitor the watercourse for signs of sedimentation during all phases of the work, undertaking or activity and take corrective action if required;
  - Maintain and clean the structures through air drying or other environmentally friendly methods;
  - Minimize physical disturbance to the site during installation, harvesting and maintenance;
  - Install and remove structures on calm days to minimize the suspension of fine sediment particles into the water column and to preferably install anchors in the winter when effects of turbidity on other organisms would be minimal.
- Install anchors in the winter when eelgrass is dormant;
- Minimize disturbance to eelgrass by sizing anchors appropriately or by installing them permanently, to prevent dragging under tension or adverse weather conditions;
- Design and install structures to maximize openings to increase light penetration;
- Develop and implement a response plan to avoid a spill of a deleterious substance:
  - Stop works, undertakings and activities in the advent of a spill of a deleterious substance;
  - Report any spills of oil, fuel or other deleterious material, whether near or directly into a waterbody;
  - Keep an emergency spill kit on site during the work, undertaking or activity;
  - Ensure clean-up measures are suitably applied to as not to result in further alteration of the watercourse;
  - Clean-up and appropriately dispose of water contaminated with deleterious substances;
  - Maintain all machinery on site in a clean condition and free of fluid leaks and aquatic invasive species;
  - Wash, refuel and service machinery and store fuel and other materials for the machinery in such a way as to prevent any deleterious substances from entering the water;
  - Refuel motors with care to prevent drips or spills. Refuelling should take place at a location where spill clean-up equipment is readily available;
  - Fuel equipment off the ice, in areas with impermeable surfaces, if conducting winter harvesting activities on the ice;
  - Use an impermeable surface (i.e. tarp or absorbent pad) if equipment must be fuelled on the ice during an emergency to assist in containing spills;
  - Use anchors that are made of clean, non-toxic material and precast and pre-cured away from the water to avoid seepage of potentially toxic substances into the waterbody;
  - Dispose operational waste (e.g. low grade oysters, non-biodegradable growing materials, shells and fouling organisms) in accordance with Provincial Regulations and/or local by-laws.
  - Replace or clean buoys, backlines, scope lines and cages regularly to avoid and minimize the build-up of bio-fouling organisms on underwater infrastructure, as well as accumulation of material onto the bottom substrate;
  - De-clump and clean shellfish in the water from where they were collected and transport them in a minimal amount of water. Spray down of equipment is recommended to minimize the movement of epifauna;
  - Clean equipment and boats that are transferred from one marine waterbody to another of any sediments, plants or animals by washing with freshwater and/or spraying with undiluted vinegar or treated with hypersaline waters, prior to being mobilized to the project site.

- Identify aquatic invasive species of concerns and report any sightings to DFO's National Aquatic Invasive Species Core Program.
- Carry out a post monitoring survey (1, 3 and 5 years), including photos, video and characterization of the eelgrass vegetation within the aquaculture lease boundary and at a reference site (outside of the lease boundary). The post monitoring survey would be part of the Environmental Monitoring Program as per requirement by the Nova Scotia Department of Fisheries and Aquaculture.

Provided that these measures are incorporated into the plans, the Program is of the view that the proposal is not likely to result in the contravention of the above-mentioned prohibitions and requirements.

Should plans change or if the proponent has omitted some information in the proposal, further review by the Program may be required. Consult our website (http://www.dfo-mpo.gc.ca/pnw-ppe/index-eng.html) or consult with a qualified environmental consultant to determine if further review may be necessary. It remains the proponent's responsibility to remain in compliance with the *Fisheries Act*, the *Species at Risk Act* and the *Aquatic Invasive Species Regulations.*

It is also the proponent's *Duty to Notify* DFO if they have caused, or are about to cause, the death of fish by means other than fishing and/or the harmful alteration, disruption or destruction of fish habitat. Such notifications should be directed to (http://www.dfo-mpo.gc.ca/pnw-ppe/contact-eng.html).

It remains the proponent's responsibility to meet all other federal, territorial, provincial and municipal requirements that apply to the proposal.

Please note that the advice provided in this letter will remain valid for a period of 1 year from the date of issuance. If the proponent plans to execute the proposal after the expiry of this letter, we recommend that he contact the Program to ensure that the advice remains up-to-date and accurate. Furthermore, the validity of the advice is also subject to there being no change in the relevant aquatic environment, including any legal protection orders or designations, during the 1-year period.

If you have any questions with the content of this letter, please contact Renelle Doucette at our Moncton office at (506) 533-6523 or by email at renelle.doucette@dfo-mpo.gc.ca. Please refer to the file number referenced above when corresponding with the Program.

Yours sincerely,

Gilles Paulin
Senior Biologist, Regulatory Reviews
Fish and Fish Habitat Protection Program

cc. Chris D Mills (DFO-Aquaculture Division)

Town Point Consulting Inc.

March 3, 2021

NS Department of Fisheries and Aquaculture
1575 Lake Road
Shelburne, NS
B0T 1W0

Attention; Nathaniel Fiendel, Lewis Clancey

Re: AQ1442,1443,1444

Please see below our reply to the Network Agency Review received from Fisheries and Oceans Canada.

Fisheries and Oceans Canada

1) Proposed lease areas are not within “Graham’s Cove”, please wording change to “Antigonish Harbour”.
2) Installation of anchors in winter will be very difficult because the sites do not freeze to safe ice thickness. Can we have a waiver on this requirement to permit anchor installation during safe conditions – spring, summer or fall?
3) De-fouling is planned to be done by dipping in heated seawater. Is it permissible to discharge the fouled, heated water into the watercourse? Are there protocols for this discharge such as tempering prior to discharge?
4) Captain’s Pond and Antigonish Harbour are connected to one another and the pond is not tidal to any other bodies of water. Are they considered to be the same body of water?

Regarding the Comments, concerns….section we have the following responses;

1) Our observation over many years indicate recreational fishers do not use the areas we have selected. They fish within the channel or further south than the selected areas. We can provide witnesses, who are among this group of users, to this fact if need be.

2) The four oyster relay fishers have met with us and we have agreed in writing to the areas we have proposed. They each have signed this agreement and we are told they will not be opposing our application.

3) The other fisheries in the harbour including eel, smelt, gaspereau all occur much further south in the harbour and are not in conflict with the proposed lease sites. Lobster is not fished within the harbour. Only silvers sides are fished near the lease areas and this fisher, Archie MacKenzie and his son, Christopher, both support our application and have said so in writing.

4) The email regarding lobster fishers concerns with navigation (attached) was not included with the response form. This is the first we have heard of any such concern even though we wrote to each of these fishers including a map of the proposed sites …we received no response. We had brief discussions with some of the fishers, no concerns were raised.

5) We have reviewed the document dated Dec 10, 2020 accompanying this response and commit to following the recommended mitigation measures associated with potential effects to fish and fish habitat.

As discussed, we would welcome a meeting with DFO to discuss the concerns they have expressed and to explore ways to mitigate these concerns.

Best regards,
Ernie Porter, P.Eng., President

Town Point Consulting Inc.

From: Clancey, Lewis <Lewis.Clancey@novascotia.ca>
Sent: Thursday, March 25, 2021 3:53 PM
To: Braid, Leanna <Leanna.Braid@dfo-mpo.gc.ca>
Cc: Dwyer, Alan <Alan.Dwyer@dfo-mpo.gc.ca>; Doucette, Renelle <Renelle.Doucette@dfo-mpo.gc.ca>; Feindel, Nathaniel J <Nathaniel.Feindel@novascotia.ca>
Subject: Reply to DFO Comments from Town Point Oysters

Hi Leanna,

As per our phone discussion earlier I've attached the reply to the DFO comments submitted to us by Town Point Oysters to be distributed to those on the call and any other pertinent DFO personnel.

I've also cc'd the original reviews (Alan Dwyer and Gilles Paulin-via Renelle Doucette) as they provided the initial DFO response.

Thanks,

Lew

Lewis Clancey

Aquaculture Development Advisor

NS Dept. of Fisheries and Aquaculture

1800 Argyle St.

WTCC 6th Floor, Suite 604,

Halifax, NS

B3J2R5

902 956 3839

lewis.clancey@novascotia.ca

Fisheries and Oceans Canada | Pêches et Océans Canada

Fisheries and Oceans Canada
Resource Management
2920 Highway 104
Antigonish County, NS
B2G 2K6

April 23, 2021

Attention; Nathaniel Fiendel, Lewis Clancey
Nova Scotia Department of Fisheries and Aquaculture

Please see DFO responses to Mr. Porter's questions embedded in the attached document.

If you have any other questions, don't hesitate to get in touch.

Thank you,

Leanna Braid

Resource Management Officer
Fisheries and Oceans Canada
Government of Canada
Gulf Nova Scotia
2920, Highway 104
Antigonish County, NS B2G 2K6
Leanna.braid@dfo-mpo.gc.ca
902-735-7130

Agent de gestion des ressources
Pêches et Océans Canada
Gouvernement du Canada
Nouvelle Écosse – Golfe
2920 route 104
Comté d'Antigonish (N.-É) B2G 2K6
Leanna.braid@dfo-mpo.gc.ca
902-735-7130

DFO Response

March 6, 2021

NS Department of Fisheries and
Aquaculture 1575 Lake Road
Shelburne, NS
B0T 1W0
Attention; Nathaniel Fiendel, Lewis Clancey

Re: AQ1442,1443,1444

Please see below our reply to the Network Agency Review received from Fisheries and Oceans Canada.

1) Proposed lease areas are not within “Graham’s Cove”, please wording change to “Antigonish Harbour”.

DFO Reply:
A note has been placed on file that “Graham’s Cove” be removed as a location descriptor.

2) We recognize that on many farms in the Gulf Region installation of anchors in winter would be preferred by the operator due to the ease of installation while working on ice. Unfortunately, the sites we have chosen do not freeze to safe conditions so working from ice is not a option in our case. Given that we will need to work from boats either in fall before the shoreline freezes or in spring after ice is gone it would be helpful to know what dates we could expect to start anchor installation in fall and end in spring.

DFO Reply:
The Fish and Fish Habitat Protection Program, has since moved to less prescriptive measures in regards to the installation of aquaculture lease anchors and the protection of eelgrass. Please consider the following related avoidance measures:

3) **Locate off-bottom aquaculture structures in areas with minimal eelgrass or vegetation cover;**
4) **Minimize disturbance to eelgrass or vegetation by sizing anchors appropriately or by installing them permanently, to prevent dragging under tension or adverse weather conditions;**

In summary, anchor installation can be undertaken year round as long the proponent considers and implements measures to avoid or minimize disturbance to eelgrass or other vegetation.

3) De-fouling is planned to be done by dipping in heated seawater. Is it permissible to discharge the fouled, heated water into the watercourse? Are there protocols for this discharge such as tempering prior to discharge?

DFO Reply:

The *Aquaculture Activities Regulations* (AAR) provide authorization, under certain conditions, to deposit biochemical oxygen demanding matter (e.g., fish food, feces, biofouling material) related to aquaculture fish production. The AAR stipulate that the owner or operator of the facility takes reasonable measures to mitigate the risk of serious harm to fish outside the facility that are part of a commercial, recreational or Aboriginal fishery.

The owner or operator is also required to submit an annual report to the Minister in accordance with section 16 of the AAR.

4) Captain's Pond and Antigonish Harbour are connected to one another and the pond is not tidal to any other bodies of water. Are they considered to be the same body of water?

DFO Reply:

Captains Pond drains into Antigonish Harbour and is hence part of the greater Antigonish Harbour watershed. However, they are considered separate bodies of water as their water sources are different.

As per Section 55 and Section 56 of the Fishery General Regulations, a licence must be required when transferring shellfish between leases/facilities. This is the case regardless of whether or not bodies of water are considered to be the same or separate.

Regarding the Comments, concerns….section we have the following responses;

7) Our observation over many years indicate recreational fishers do not use the areas we have selected. They fish within the channel or further south than the selected areas. We can provide witnesses, who are among this group of users, to this fact if need be.

8) The four oyster relay fishers have met with us and we have agreed in writing to the areas we have proposed. They each have signed this agreement and we are told they will not be opposing our application.

9) The other fisheries in the harbour including eel, smelt, gaspereau all occur much further south in the harbour and are not in conflict with the proposed lease sites. Lobster is not fished within the harbour. Only silvers sides are fished near the lease areas and this fisher, Archie MacKenzie and his son, Christopher, both support our application and have said so in writing.

10)We have reviewed the document dated Dec 10, 2020 accompanying this response and commit to following the recommended mitigation measures associated with potential effects to fish and fish habitat.

As discussed, we would welcome a meeting with DFO to discuss the concerns they have expressed and to explore ways to mitigate these concerns.

Best regards, Ernie Porter, P.Eng., President Town Point Consulting Inc.

Antigonish – Preliminary Eelgrass Survey – Draft Report

Jeff Barrell
DFO-Gulf Region
9/24/2019

Preliminary eelgrass data were collected targeting prospective shellfish aquaculture leases in Antigonish Harbour, NS, on 20 September 2019. Data were collected by Jeff Barrell and Venitia Joseph, both with DFO Science in Gulf Region. The survey consisted of single-beam sonar data collected at three potential lease areas within the bay. Additional imagery was collected for ground-truthing (i.e. underwater videos) and aerial mapping (i.e. drone survey).

Survey methods:

- The survey was relatively short due to the travel time from Moncton to Antigonish (~3 hours).
- All work was conducted from a 17 ft Boston Whaler launched at the Antigonish Boat Club.
- Single-beam sonar (Biosonics Habitat-MX, 208 kHz, 8.6° beam angle):
    - Differential GPS positioning was recorded for all sonar data.
    - Sonar transects were planned to be separated by 50-100 meters, though this varied in practice.
    - Bathymetric and eelgrass data were produced through a combination of automated and manual echogram classification using Biosonics Visual Habitat software; further analyses conducted with ArcGIS v10.7.
    - Eelgrass data are represented either as raw presence/absence on a ping-by-ping basis, or as summary reports averaging 10 pings. In the latter instance, eelgrass is quantified as percent cover, calculated by:
      (number of vegetated pings) / (total number of pings per report) x 100
    - Eelgrass canopy height detection threshold was set at 15 cm, meaning that shoot height must exceed the stated threshold to be identified as a vegetated ping.
- Drone imagery was collected using a DJI Phantom 4 Pro+ v2.0 UAV. Given the short amount of time remaining for aerial imagery, only the outer area (proposed lease areas #3, Figure 1) was surveyed. The UAV was flown manually at approximately 90 meters above ground level. Nadir-facing imagery was collected.
- Underwater video for ground-truthing was collected at a limited number of points for validation and ground-truthing of remote sensing data. Videos were collected using a pole-mounted GoPro camera.

Notes:

- Having not previously worked in this bay, navigation was slower than expected, leaving less time for drone imagery than was ideal, only allowing for a survey of the outer area. The timing of imagery was relatively poor with respect to sun angles (i.e. mid-afternoon) and tides (high tide was at 3:28 pm). Survey was flown manually, so will likely be of limited use for explicit geolocation, though adequate for a qualitative assessment of patchiness.
- Survey took place approximately one week after a major storm event (hurricane/tropical storm Dorian); effects on the eelgrass community are unknown, but likely not severe, as the canopy seemed more or less intact, and there was a relatively normal level of wracking evident.
- Subjectively, a relatively large amount of eelgrass was present throughout the estuary, outside of deeper channels, though seemingly with a patchy fragmented distribution over most areas. Some areas exhibited moderately high epiphyte loads that were visible from drone imagery.
- Evidence of green crab damage was present (i.e. wrack shoots that had been stripped consistent with green crab behaviour).
- Aside from eelgrass, some other macrophytes were present at low densities, including likely *Fucus* and other macroalgae. This was only observed visually and was not widespread.
- Drone imagery shows what appears to be wild shellfish beds (likely blue mussel) present in some areas; this merits further investigation.
- Due to the above points, imagery must be interpreted with care, and likely a more detailed aerial survey combined with more intensive and quantitative ground-truthing (i.e. drop camera with frame) would be beneficial and complementary to acoustic data.

Biosonics:

- The early part of the survey, covering the southernmost potential lease area, was truncated due to shallow water; this part of the survey began near to the lowest tide, and was shallow to begin with. This area would likely be conducive to drone surveys due to depth.
- The most recent proposed lease polygons provided by the province include some areas that are too shallow for access, particularly in area #3; polygons may have been created using an outdated coastline file, and may require adjustment.

Suggestions:

- Likely some value in a more detailed UAV survey flown with an autopilot to ensure full coverage with sufficient overlap for image analysis.

- In conjunction, conduct more extensive ground-truthing with underwater photos/videos, using preliminary images and acoustic data to identify areas of interest (e.g. macrophytes other than eelgrass, wild shellfish)

Results:

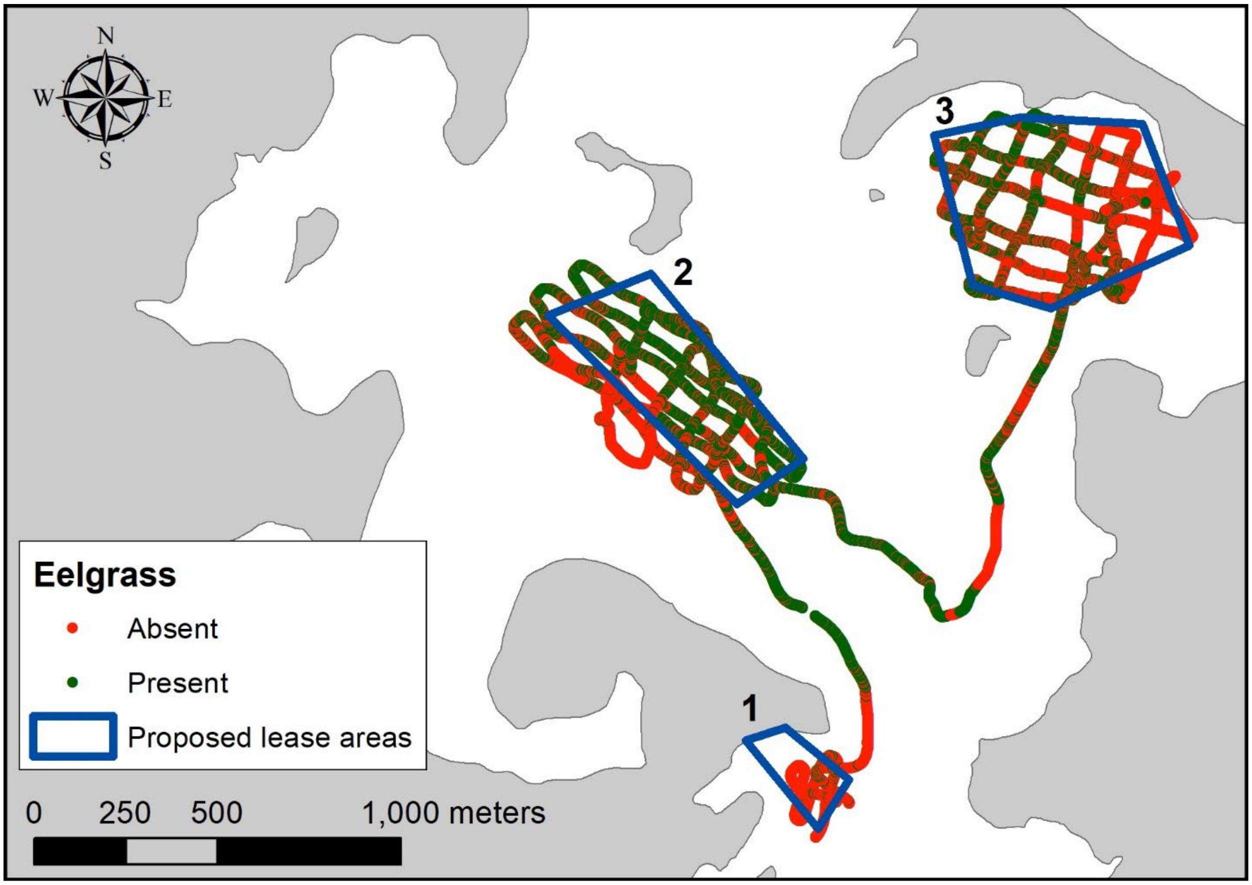


Figure 1. Overview of three proposed lease areas with overlay of sonar tracks; categorized on a single-ping basis as eelgrass presence or absence.

Eelgrass presence-absence is depicted in **Figure 1** on a per-ping basis; note that due to the relatively high ping rate (5 Hz), these data points overlap and likely obscure patterns when viewed over a broad extent. Despite this, general patterns can be observed in the results showing significantly more “absence” in area #3 when compared to area #2. A large proportion of absences were found in area #1 as well, though much of this area was inaccessible due to shallow depths.

Aquatic vegetation was easily identified through echogram analysis at this site; the seafloor generally consists of homogeneously distributed soft sediments with little bathymetric variation, providing ideal contrast for vegetation detection (**Figure 2**). The eelgrass community in the surveyed areas consists of a range of densities, with areas of near-continuous cover (e.g. **Figure 2a**) as well as regions with very sparse or patchy cover (e.g. **Figure 2b**) and occasional bare patches. As expected, eelgrass was absent from deeper parts of the navigation channel, and was found at a maximum depth of 2.76 m (relative to water depth at time of survey, unadjusted for tide).

While eelgrass occurred in some form throughout most of the surveyed area, it exhibited heterogeneity and patchiness at all areas, rarely forming dense agglomerations that could be categorized as “continuous” beds. Patch gaps were apparent from visual inspection as well as underwater video (**Figure 3**) and drone imagery. There were also apparent intracanopy gaps that were not easily detected by sonar; this may merit further investigation of the data. These gaps may be explained by the timing of the survey (i.e. relatively late in the season, and shortly after a large storm), rather than representative of the landscape at large, though this is unknown and would require a return visit in the growing season for assessment.

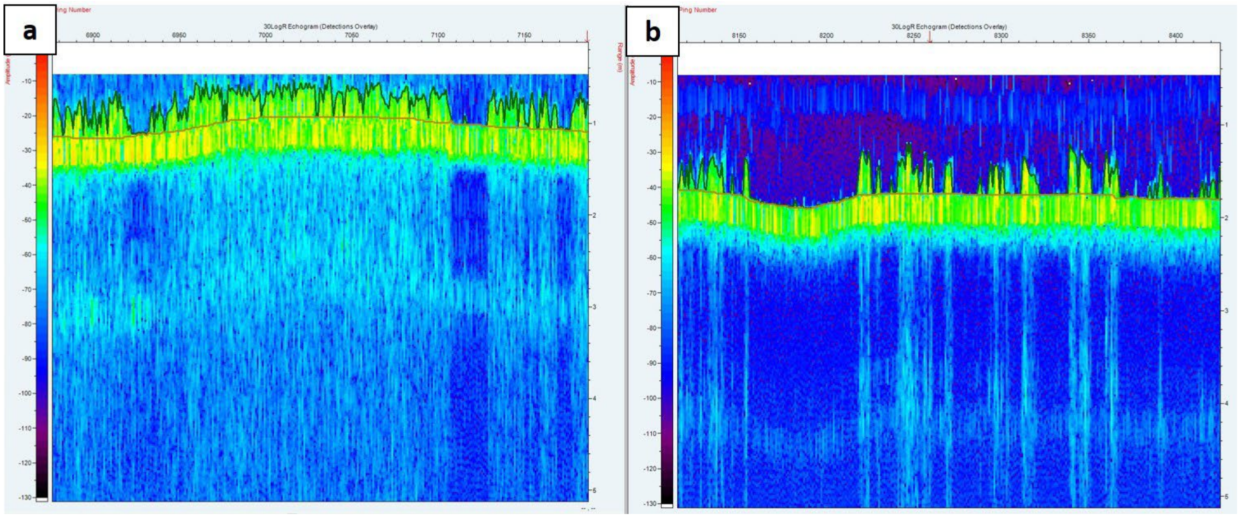


Figure 2. Example echograms showing a) dense and b) sparse eelgrass coverage. Each echogram consists of approximately 300 pings moving along a single transect and represents a cross-sectional view of the water column, with depth on the x-axis. Brown line indicates the seafloor while the green line identifies the plant canopy. Both transects are from proposed area #3.

Figure 3. Screenshots of underwater video collected to validate eelgrass presence. Upper-left photo is from area #3; other photos all from area #2. Note range of densities and epiphyte loads.

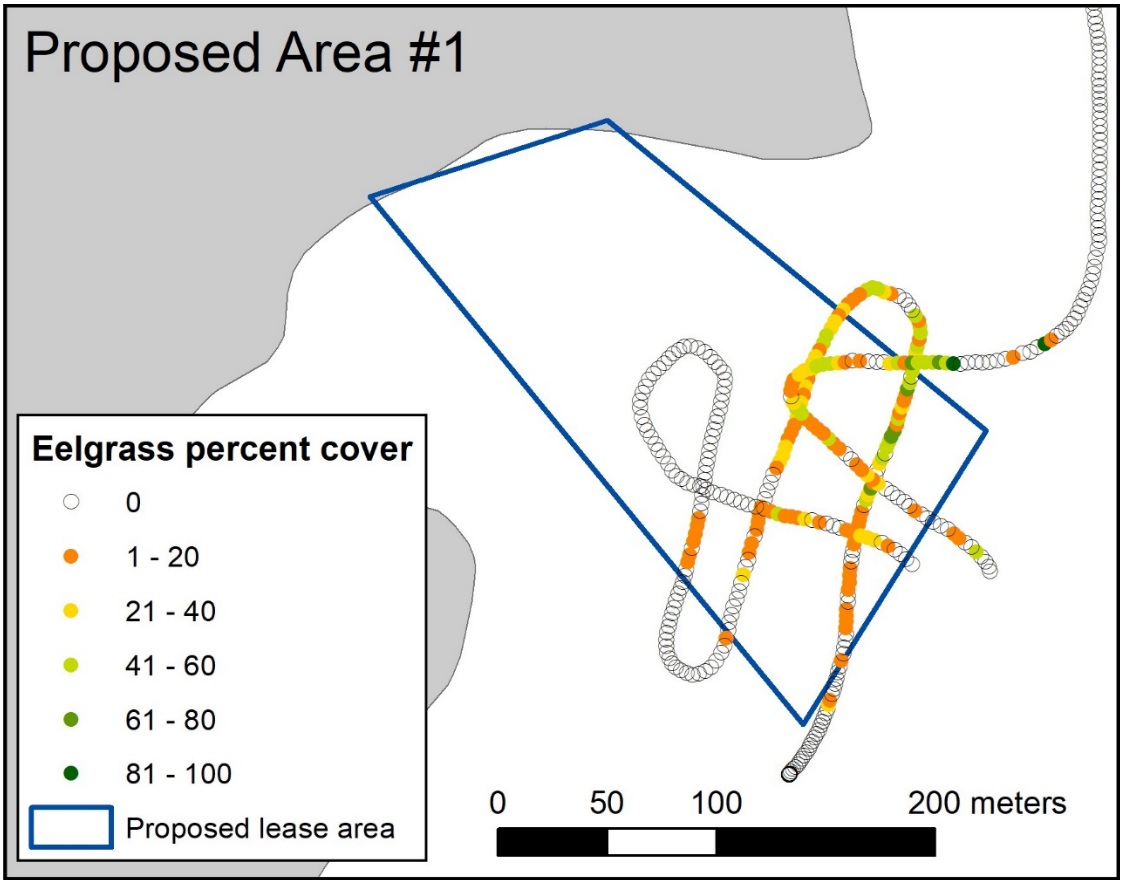


Figure 4. Sonar results for proposed area #1; each data point represents the proportion of vegetated pings over a 10-ping cycle.

Data collected for area #1 were insufficient for detailed mapping due to depth limitations. However, there were notably large gaps in eelgrass occurrence over the surveyed area, as can be seen in the results (**Figure 4**). Unfiled circles each represent a series of 10 pings without any vegetation, and very few of the output points exceeded 50% cover. This is not unexpected given the site's proximity to a dock/wharf as well as its sheltered and shallow setting with apparently very soft sediments.

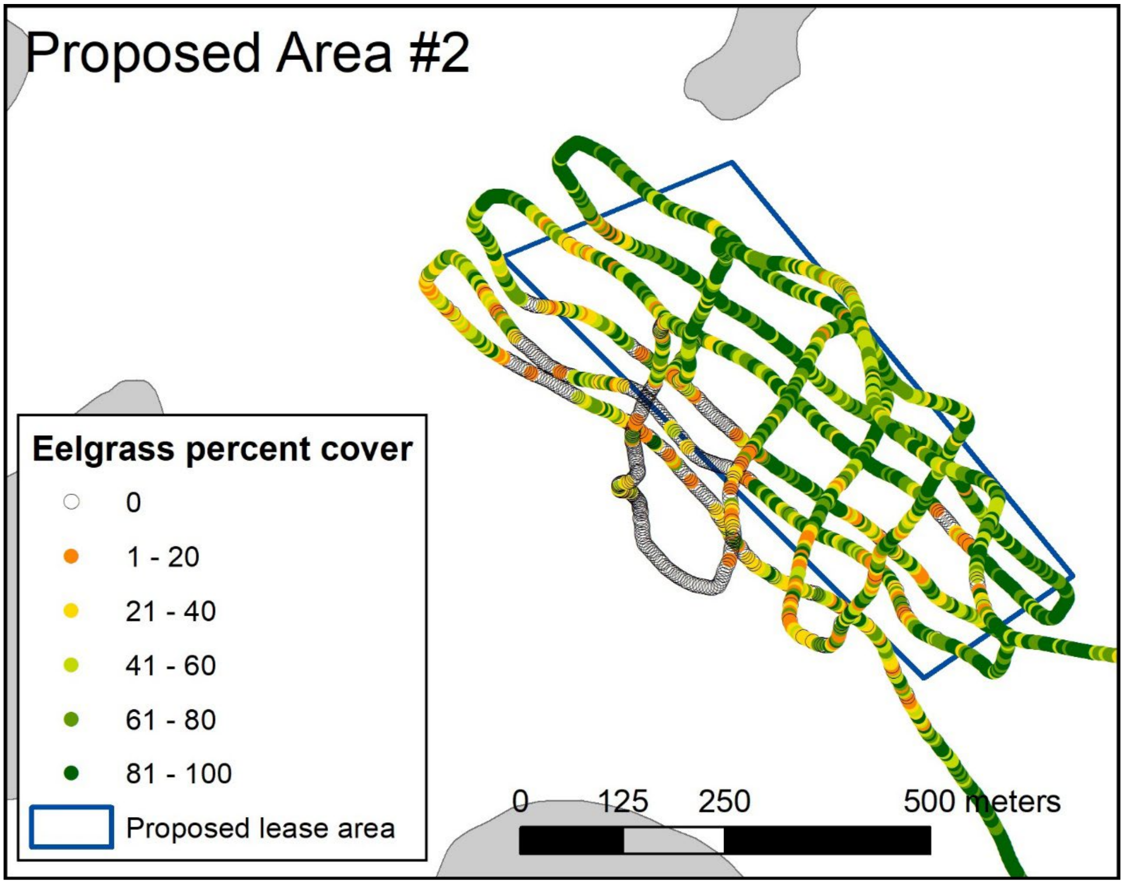


Figure 5. Sonar results for proposed area #2; each data point represents the proportion of vegetated pings over a 10-ping cycle.

Proposed are #2 contained the largest quantity and density of eelgrass of the three surveyed areas (**Figure 5**). Eelgrass occurred, even if only at low density, along nearly all survey transects, although notably there seemed to be areas of absence immediately to the west of the site boundary. This site borders the inlet channel to the east, and shallows approaching the channel as well as the island to the north; the transects shown delineate the navigable area at time of survey. Eelgrass density seemed to increase with decreasing depth, consistent with scientific understanding of eelgrass habitat (e.g. balance between shelter from hydrodynamic forces and sufficient flow).

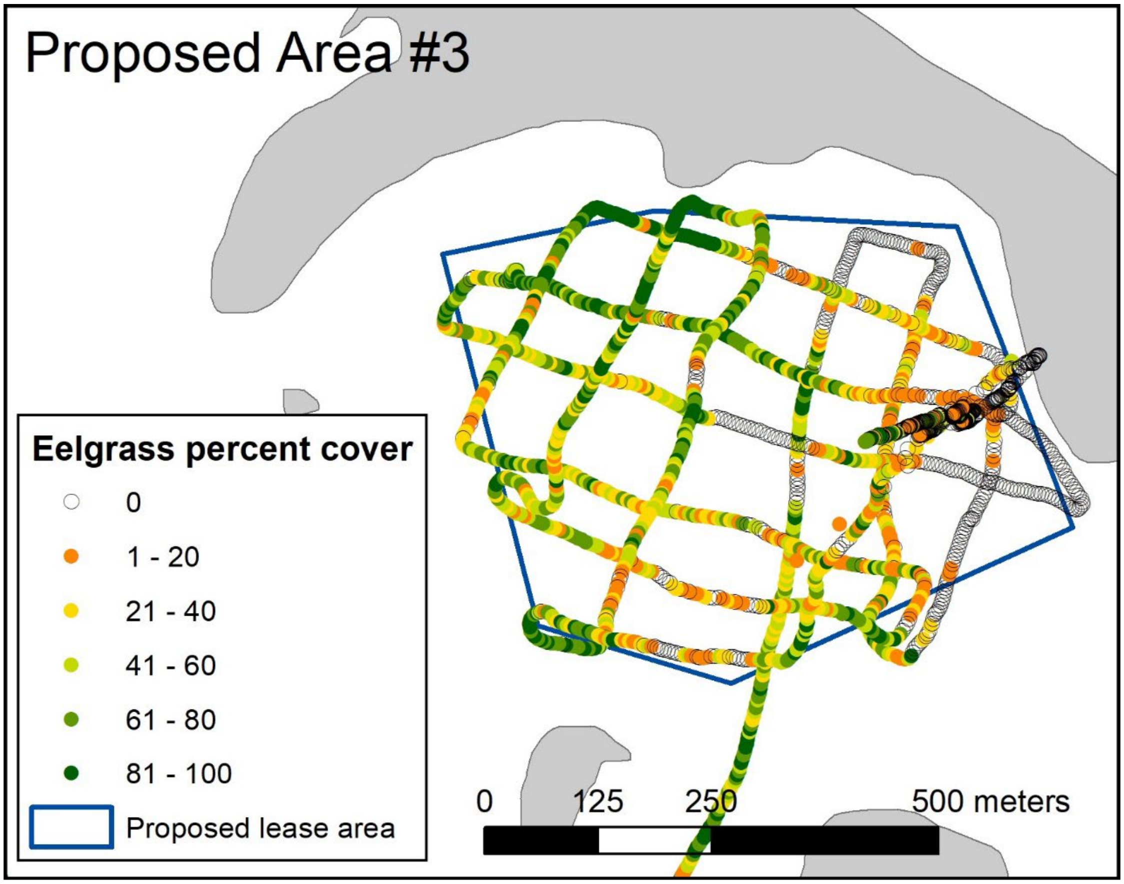


Figure 6. Sonar results for proposed area #3; each data point represents the proportion of vegetated pings over a 10-ping cycle.

In contrast to area #2, area #3 showed a distinct pattern in eelgrass occurrence, with very low density to the east of the site increasing towards the west (**Figure 6**). Large areas of absence occur, particularly to the east and in the central portion of the site. Eelgrass that does occur was patchy and fragmented in most areas except those most sheltered by the barrier beach to the north and the island to the south.

This heterogeneity can be seen in the drone imagery collected at the site (**Figure 7**). Although this imagery cannot easily be used for detailed mapping, it clearly depicts the fragmented nature of eelgrass habitat in this area.

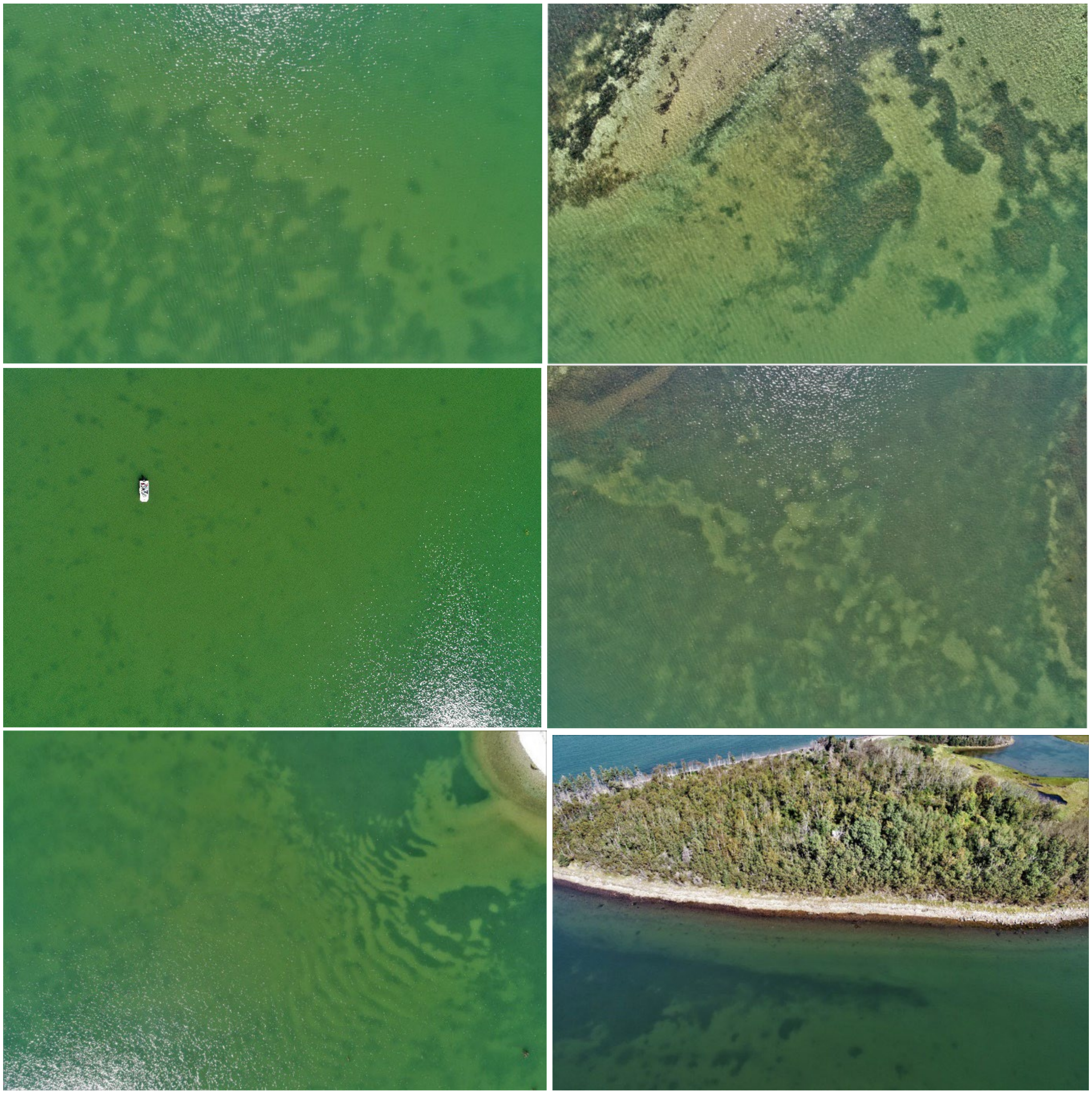


Figure 7. Collection of UAV images collected over proposed lease area #3. Darker subtidal areas are most likely eelgrass, showing a patchy fragmented distribution mostly, with some exceptions closer to land and in sheltered areas.

APPENDIX B: CANADIAN FOOD INSPECTION AGENCY

From: Greenwood, Megan N
Sent: October 7, 2020 2:26 PM
To: chris.mills@dfo-mpo.gc.ca; Braid, Leanna <Leanna.Braid@dfo-mpo.gc.ca>; Smith, Angela (CFIA/ACIA) <angela.smith@canada.ca>; david.macarthur@ec.gc.ca; Birch, Angela <Angela.Birch@novascotia.ca>; Kittilsen, Michael A <Michael.Kittilsen@novascotia.ca>; Cottreau-Robins, Catherine M <Catherine.Cottreau-Robins@novascotia.ca>; Blackburn, Lori M <Lori.Blackburn@novascotia.ca>; Boudreau, Louise O <Louise.Boudreau@novascotia.ca>; Gautreau, Rachel (EC) <rachel.gautreau@canada.ca>
Cc: Feindel, Nathaniel J <Nathaniel.Feindel@novascotia.ca>; Clancey, Lewis <Lewis.Clancey@novascotia.ca>; Ceschiutti, Robert <Robert.Ceschiutti@novascotia.ca>; King, Matthew S <Matthew.King@novascotia.ca>; Hancock, Bruce H <Bruce.Hancock@novascotia.ca>
Subject: AQ#1442, 1443, 1444 - New Application

Attention: Network Review Agencies

Please see the attached memo for three (3) Aquaculture Application Nos. 1442, 1443, 1444 in Antigonish Harbour, Antigonish County.

Please respond with your feedback by **December 7, 2020.**

The attached zipped shapefile requires specific geospatial software to open and is intended for convenient use by GIS professionals only.

Megan Greenwood

Licensing Coordinator
Nova Scotia Department of Fisheries and Aquaculture
1575 Lake Road
Shelburne, NS
B0T 1W0
Phone 902-875-7443
Fax 902-875-7429

Network Agency Review of an Aquaculture Application

Agency	**CFIA**
Division (if applicable)	
Date	**November 27, 2020**
File No.	**1444**
Type of application	**New Marine Shellfish**
Information Provided	

Please provide comments, concerns, recommendations, or requirements on the above stated application for a marine aquaculture licence. Please include the criterion /criteria within your jurisdiction or mandate that your feedback is based upon. Similarly, if additional information is required to make a determination, please include the criterion /criteria within your jurisdiction or mandate that your request is based upon.

☒ No concerns regarding the proposed development
☐ Concerns with development are expressed below
☐ Request modifications to the proposed development (described below)
☐ Required or recommended conditions (described below)
☐ Request additional information (described below)
☐ Request meeting with applicant and NSDFA (described below)
☐ No comments on the application

Comments, concerns, recommendations, and/or required conditions including the criterion /criteria within your jurisdiction or mandate that your feedback is based upon. (Attach comments if preferred, or add additional pages, as required.):

As with any suspended aquaculture of oysters section 9.1.3 of the CSSP manual must be adhered to.

9.1.3 Aquaculture methods

Proponents are to culture shellfish in a manner that will ensure they are safe for consumption before harvesting them for sale. If a shellfish control authority determines that the technology used to grow shellfish could potentially create or attract significant sources of contamination, failing to develop adequate control measures could lead to the aquaculture site being closed. Any shellfish cultured using this type of technology must be subject to preventive controls by a licensed operator, or the leaseholder must submit a harvest plan with appropriate control measures to the regional shellfish control authority.

Public Notice and Disclosure

As part of the process for deciding on an application, it may be necessary for the Nova Scotia Department of Fisheries and Aquaculture (“Fisheries and Aquaculture”) to disclose the collected network review information to the applicant and other government bodies, including, if applicable, the Nova Scotia Aquaculture Review Board for use at an adjudicative hearing relating to the application in question.

In accordance with departmental policy, which seeks to promote public involvement in the process for deciding on aquaculture applications, Fisheries and Aquaculture will disclose aquaculture application information, including network review information, on the departmental website.

Privacy Statement

The network review information collected as part of an aquaculture application will only be used or disclosed by Fisheries and Aquaculture for the purpose of deciding on the application.

All application information collected is subject to the Freedom of Information and Protection of Privacy Act (“FOIPOP”) and will only be used or disclosed in accordance with FOIPOP.

APPENDIX C: TRANSPORT CANADA

From: Greenwood, Megan N
Sent: October 7, 2020 2:26 PM
To: chris.mills@dfo-mpo.gc.ca; Braid, Leanna <Leanna.Braid@dfo-mpo.gc.ca>; Smith, Angela (CFIA/ACIA) <angela.smith@canada.ca>; david.macarthur@ec.gc.ca; Birch, Angela <Angela.Birch@novascotia.ca>; Kittilsen, Michael A <Michael.Kittilsen@novascotia.ca>; Cottreau-Robins, Catherine M <Catherine.Cottreau-Robins@novascotia.ca>; Blackburn, Lori M <Lori.Blackburn@novascotia.ca>; Boudreau, Louise O <Louise.Boudreau@novascotia.ca>; Gautreau, Rachel (EC) <rachel.gautreau@canada.ca>
Cc: Feindel, Nathaniel J <Nathaniel.Feindel@novascotia.ca>; Clancey, Lewis <Lewis.Clancey@novascotia.ca>; Ceschiutti, Robert <Robert.Ceschiutti@novascotia.ca>; King, Matthew S <Matthew.King@novascotia.ca>; Hancock, Bruce H <Bruce.Hancock@novascotia.ca>
Subject: AQ#1442, 1443, 1444 - New Application

Attention: Network Review Agencies

Please see the attached memo for three (3) Aquaculture Application Nos. 1442, 1443, 1444 in Antigonish Harbour, Antigonish County.

Please respond with your feedback by **December 7, 2020.**

The attached zipped shapefile requires specific geospatial software to open and is intended for convenient use by GIS professionals only.

Megan Greenwood

Licensing Coordinator
Nova Scotia Department of Fisheries and Aquaculture
1575 Lake Road
Shelburne, NS
B0T 1W0
Phone 902-875-7443
Fax 902-875-7429

May 1, 2019 1:00 p.m. - 3:00 p.m.
Antigonish Beech Hill Road Provincial Office (Conference Call info to be provided)

-----Original Appointment-----
From: Goreham, Brennan CD <Brennan.Goreham@novascotia.ca>
Sent: April 11, 2019 8:44 PM
To: Goreham, Brennan CD; Feindel, Nathaniel J; Reid, Gregor Kyle; Heighton, Ralph; Greenwood, Megan N; Hudson, Jolene; LeBlanc, Mélanie
Subject: Meeting to discuss AQ#1424 with Proponent
When: May 1, 2019 1:00 PM-3:00 PM (UTC-04:00) Atlantic Time (Canada).
Where: Antigonish Beech Hill Road Provincial Office (Conference Call info to be provided)

This meeting is to discuss Ernie Porter's (Town Point Consulting) Option to Lease (attached). DFO and TC to also be invited.

Nate/Gregor, I've booked Shelburne boardroom also.

APPENDIX D: ENVIRONMENT AND CLIMATE CHANGE CANADA – CANADIAN SHELLFISH WATER CLASSIFICATION PROGRAM

From: Greenwood, Megan N
Sent: October 7, 2020 2:26 PM
To: chris.mills@dfo-mpo.gc.ca; Braid, Leanna <Leanna.Braid@dfo-mpo.gc.ca>; Smith, Angela (CFIA/ACIA) <angela.smith@canada.ca>; david.macarthur@ec.gc.ca; Birch, Angela <Angela.Birch@novascotia.ca>; Kittilsen, Michael A <Michael.Kittilsen@novascotia.ca>; Cottreau-Robins, Catherine M <Catherine.Cottreau-Robins@novascotia.ca>; Blackburn, Lori M <Lori.Blackburn@novascotia.ca>; Boudreau, Louise O <Louise.Boudreau@novascotia.ca>; Gautreau, Rachel (EC) <rachel.gautreau@canada.ca>
Cc: Feindel, Nathaniel J <Nathaniel.Feindel@novascotia.ca>; Clancey, Lewis <Lewis.Clancey@novascotia.ca>; Ceschiutti, Robert <Robert.Ceschiutti@novascotia.ca>; King, Matthew S <Matthew.King@novascotia.ca>; Hancock, Bruce H <Bruce.Hancock@novascotia.ca>
Subject: AQ#1442, 1443, 1444 - New Application

Attention: Network Review Agencies

Please see the attached memo for three (3) Aquaculture Application Nos. 1442, 1443, 1444 in Antigonish Harbour, Antigonish County.

Please respond with your feedback by **December 7, 2020.**

The attached zipped shapefile requires specific geospatial software to open and is intended for convenient use by GIS professionals only.

Megan Greenwood

Licensing Coordinator
Nova Scotia Department of Fisheries and Aquaculture
1575 Lake Road
Shelburne, NS
B0T 1W0
Phone 902-875-7443
Fax 902-875-7429

Network Agency Review of an Aquaculture Application

Agency	**ECCC**
Division (if applicable)	**SWCP**
Date	**February 1, 2021**
File No.	**1444**
Type of application	**New Marine Shellfish**
Information Provided	

Please provide comments, concerns, recommendations, or requirements on the above stated application for a marine aquaculture licence. Please include the criterion /criteria within your jurisdiction or mandate that your feedback is based upon. Similarly, if additional information is required to make a determination, please include the criterion /criteria within your jurisdiction or mandate that your request is based upon.

☒ No concerns regarding the proposed development
☐ Concerns with development are expressed below
☐ Request modifications to the proposed development (described below)
☐ Required or recommended conditions (described below)
☐ Request additional information (described below)
☐ Request meeting with applicant and NSDFA (described below)
☐ No comments on the application

Comments, concerns, recommendations, and/or required conditions including the criterion /criteria within your jurisdiction or mandate that your feedback is based upon. (Attach comments if preferred, or add additional pages, as required.):

The area is classified as Restricted.

Public Notice and Disclosure

As part of the process for deciding on an application, it may be necessary for the Nova Scotia Department of Fisheries and Aquaculture (“Fisheries and Aquaculture”) to disclose the collected network review information to the applicant and other government bodies, including, if applicable, the Nova Scotia Aquaculture Review Board for use at an adjudicative hearing relating to the application in question.

In accordance with departmental policy, which seeks to promote public involvement in the process for deciding on aquaculture applications, Fisheries and Aquaculture will disclose aquaculture application information, including network review information, on the departmental website.

Privacy Statement

The network review information collected as part of an aquaculture application will only be used or disclosed by Fisheries and Aquaculture for the purpose of deciding on the application.

All application information collected is subject to the Freedom of Information and Protection of Privacy Act (“FOIPOP”) and will only be used or disclosed in accordance with FOIPOP.

APPENDIX E: ENVIRONMENT AND CLIMATE CHANGE CANADA – CANADIAN WILDLIFE SERVICE

From: Greenwood, Megan N
Sent: October 7, 2020 2:26 PM
To: chris.mills@dfo-mpo.gc.ca; Braid, Leanna <Leanna.Braid@dfo-mpo.gc.ca>; Smith, Angela (CFIA/ACIA) <angela.smith@canada.ca>; david.macarthur@ec.gc.ca; Birch, Angela <Angela.Birch@novascotia.ca>; Kittilsen, Michael A <Michael.Kittilsen@novascotia.ca>; Cottreau-Robins, Catherine M <Catherine.Cottreau-Robins@novascotia.ca>; Blackburn, Lori M <Lori.Blackburn@novascotia.ca>; Boudreau, Louise O <Louise.Boudreau@novascotia.ca>; Gautreau, Rachel (EC) <rachel.gautreau@canada.ca>
Cc: Feindel, Nathaniel J <Nathaniel.Feindel@novascotia.ca>; Clancey, Lewis <Lewis.Clancey@novascotia.ca>; Ceschiutti, Robert <Robert.Ceschiutti@novascotia.ca>; King, Matthew S <Matthew.King@novascotia.ca>; Hancock, Bruce H <Bruce.Hancock@novascotia.ca>
Subject: AQ#1442, 1443, 1444 - New Application

Attention: Network Review Agencies

Please see the attached memo for three (3) Aquaculture Application Nos. 1442, 1443, 1444 in Antigonish Harbour, Antigonish County.

Please respond with your feedback by **December 7, 2020.**

The attached zipped shapefile requires specific geospatial software to open and is intended for convenient use by GIS professionals only.

Megan Greenwood

Licensing Coordinator
Nova Scotia Department of Fisheries and Aquaculture
1575 Lake Road
Shelburne, NS
B0T 1W0
Phone 902-875-7443
Fax 902-875-7429

From: Greenwood, Megan N <Megan.Greenwood@novascotia.ca>
Sent: November 24, 2020 10:49 AM
To: chris.mills@dfo-mpo.gc.ca; Braid, Leanna <Leanna.Braid@dfo-mpo.gc.ca>; Smith, Angela (CFIA/ACIA) <angela.smith@canada.ca>; MacArthur, David (EC) <david.macarthur@canada.ca>; Birch, Angela <Angela.Birch@novascotia.ca>; Cottreau-Robins, Catherine M <Catherine.Cottreau-Robins@novascotia.ca>; Blackburn, Lori M <Lori.Blackburn@novascotia.ca>; Boudreau, Louise O <Louise.Boudreau@novascotia.ca>; Gautreau, Rachel (EC) <rachel.gautreau@canada.ca>
Cc: Clancey, Lewis <Lewis.Clancey@novascotia.ca>; Ceschiutti, Robert Robert.Ceschiutti@novascotia.ca

Subject: FW: AQ#1442, 1443, 1444 - New Application
Attn: Network Review Agencies:
Please be reminded that our office has not received comments from your Department for the proposed aquaculture site in Antigonish Harbour, Antigonish County. Your comments are due on or before **December 7, 2020.**

Megan Greenwood
Licensing Coordinator
Nova Scotia Department of Fisheries and Aquaculture
1575 Lake Road
Shelburne, NS
B0T 1W0
Phone 902-875-7443
Fax 902-875-7429

From: Gautreau, Rachel (EC)
To: Greenwood, Megan N
Cc: Mailhiot, Joshua (EC)
Subject: RE: AQ#1442, 1443, 1444 - New Application
Date: December 7, 2020 3:05:20 PM
Attachments: Re Aquaculture Lease Application for Antigonish Harbour.msg
Aquaculture Lease Application for Antigonish Harbour.msg
RE Aquaculture Lease Application for Antigonish Harbour.msg

Hi Megan,
I’m currently working on our comments for these applications, and hope to get comments to you asap, but wanted to also make you aware that the Friends of Antigonish Harbour have been emailing me regarding these sites. Attached for your information are the two emails that I received from them, as well as my response to their first email.
Thanks,
Rachel

From:	Gautreau, Rachel (EC)
To:	Greenwood, Megan N
Cc:	Mailhiot, Joshua (EC); Hanson, Al (EC)
Subject:	Proposed aquaculture application AQ#1444 - Antigonish Harbour, N.S.
Date:	December 7, 2020 4:25:00 PM

Hi Megan,

Environment and Climate Change Canada's Canadian Wildlife Service (CWS) has reviewed the information forwarded to us regarding proposed aquaculture application AQ#1444 in Antigonish Harbour, Nova Scotia. The proposed lease is located less than 300 m from Dunn's Beach sandspit which is critical habitat for Piping Plovers and has been used by a nesting tern colony. The proposed lease is also less than 300 m from Gooseberry Island, a nesting island for 2 species of gulls. **We recommend that proposed lease AQ#1444 be relocated to an alternate location at least 300 m from all areas of Dunn's Beach Sandspit and Gooseberry Island**.

Piping Plovers and breeding seabirds are particularly vulnerable to the effects of human disturbance. The period prior to egg-laying is very important as this is when they engage in pair formation and other important breeding behaviours, such as nest site defense, nest building, and copulation. Disturbance prior to egg-laying may cause birds to abandon historical nesting locations. Meanwhile, disturbance during the breeding season can cause these birds to abandon their nests or young, or to use valuable energy reserves for defence, instead of incubating eggs and feeding their young. The presence of humans in close proximity to nests may prevent parent birds from returning to protect and feed their young, and expose eggs or chicks to predation, and to the lethal effects of heat, cold and rain. When parent birds are flushed, young chicks may wander from their nest site and be taken by predators, or seabird chicks may be pecked to death by neighbouring birds.

For avoidance of disturbance to colonial nesting waterbirds and Piping Plovers, CWS generally recommends the following:

- In general, maintain a minimum distance of at least 300 m from all areas of the island or colony occupied by seabirds and waterbirds, and Piping Plover beaches.
- For high-disturbance activities (e.g. drilling, blasting), maintain a buffer of at least 1 km from colonies.

It is also extremely important that mitigation measures, such as the following, be implemented to avoid/minimize adverse effects on migratory birds, including Piping Plovers and colonial nesters at a relocated lease AQ#1444:

- Boats and equipment should stay at least 300 m from Dunn's Beach sandspit and Gooseberry Island during spring and summer, and marine travel in the vicinity should take place at steady speeds, moving parallel to the shore, rather than approaching the sandspit or island directly.

- Vessels and equipment should be well muffled, and the proponent/contractors should avoid any sharp or loud noises, should not blow horns or whistles, and should maintain constant engine noise levels. Due to the proximity to sensitive receptors, we recommend replacing whistle blasts and horns with radio communications.

- Marine vessels should not pursue seabirds/waterbirds swimming on the water surface, and avoid concentrations of birds on the water.

- Oil or waste should never be dumped overboard, as even small amounts of oil can kill birds and other marine life, and habitats may take years to recover.

- There should be no access to the Dunn's Beach sandspit or Gooseberry Island, including the intertidal zone, by project staff and/or equipment. Should equipment wash up at this site during spring or summer, the proponent would be expected to contact CWS and provincial wildlife biologists to ensure that Piping Plovers and colonial nesters are not disturbed during retrieval of equipment and to ensure compliance with the *Species at Risk Act* (SARA), the *Migratory Birds Convention Act* (MBCA), and provincial wildlife legislation. CWS and provincial wildlife biologists may restrict access to some areas during sensitive periods.

- Food scraps and other garbage left on beaches and other coastal habitat can artificially enhance the populations of avian and mammalian predators of eggs and chicks of Piping Plovers and colonial nesters. No litter (including food scraps) should be left in coastal areas.

- Since even small spills of oil can have very serious effects on birds, every effort should be taken to ensure that not oil spills occur. The proponent should ensure that all precautions are taken by staff to prevent fuel leaks from equipment, and contingency plans in case of oil spills should be prepared.

- Project staff and vessels should not approach concentrations of seabirds, waterfowl or shorebirds.

- The proponent should ensure that staff/contractors are familiar with all mitigation measures and are prepared to implement these. In the event of a discrepancy between environmental legislation and these measures, the requirements of the legislation will take precedence.

- Common Eider chicks are known to be very susceptible to depredation by gulls, and activities such as boat traffic, that could separate family groups could have disastrous consequences for the fledging success of these birds.

  It is extremely important that project activities do not disturb female eiders with broods of chicks, groups of moulting eiders, or flocks of staging or wintering birds. Contractor/staff should not approach concentrations of seabirds, waterfowl, or shorebirds; and have well muffled vessels and equipment. Careful planning of access routes is very important, and vessels should reduce speed when in the vicinity of flocks of birds.

- Contamination problems due to bird feces have occurred at aquaculture sites using vexar bags. Birds in some areas have been observed perching on vexar bags. Other birds may be attracted to fouling organisms as a potential food source. The placement of vexar bags in any area of high bird use could result in the contamination of cultured shellfish.

  If project proponents are allowed to proceed with an aquaculture site in an area of high bird use, then they should be advised that they do so at their own risk. Migratory birds are all protected under the *Migratory Birds Convention Act* and associated regulations (MBCA), and under this act, it is illegal to kill, take, or hunt any species of migratory bird or to destroy nests or young of a migratory bird without a permit. Under the *Migratory Birds Regulations*, hunt "means chase, pursue, worry, follow after or on the trail of, lie in wait for, or attempt in any manner to capture, kill, injure or harass a migratory bird, whether or not the migratory bird is captured, killed or injured".

Other expert departments

We also recommend that provincial wildlife and wetlands biologists be given the opportunity to review this project proposal if this has not already occurred.

Applicable Legislation

The *Migratory Birds Convention Act* (MBCA) protects most bird species in Canada however, some families of birds are excluded. A list of species under MBCA protection can be found at https://www.canada.ca/en/environment-climate-change/services/migratory-birds-legal-protection/list.html .

Under Section 6 of the *Migratory Birds Regulations* (MBR), no person shall disturb, destroy or take a nest or egg of a migratory bird; or to be in possession of a live migratory bird, or its carcass, skin, nest or egg, except under authority of a permit. It is important to note that under the current MBR, no permits can be issued for the incidental take of migratory birds caused by development projects or other economic activities. Furthermore, Section 5.1 of the MBCA describes prohibitions related to deposit of substances harmful to migratory birds:

> “5.1 (1) No person or vessel shall deposit a substance that is harmful to migratory birds, or permit such a substance to be deposited, in waters or an area frequented by migratory birds or in a place from which the substance may enter such waters or such an area.
>
> (2) No person or vessel shall deposit a substance or permit a substance to be deposited in any place if the substance, in combination with one or more substances, results in a substance — in waters or an area frequented by migratory birds or in a place from which it may enter such waters or such an area — that is harmful to migratory birds.”

It is the responsibility of the proponent to ensure that activities comply with the MBCA and regulations. In fulfilling its responsibility for MBCA compliance, the proponent should take the following points into consideration:

- Information regarding regional nesting periods can be found at https://www.canada.ca/en/environment-climate-change/services/avoiding-harm-migratory-birds/general-nesting-periods.html.. Some species protected under the MBCA may nest outside these timeframes

- Most migratory bird species construct nests in trees (sometimes in tree cavities) and shrubs, but several species nest at ground level (e.g., Common Nighthawk, Killdeer, sandpipers), in hay fields, pastures or in burrows. Some bird species may nest on cliffs or in stockpiles of overburden material from mines or the banks of quarries. Some migratory birds (including certain waterfowl species) may nest in head ponds created by beaver dams. Some migratory birds (e.g., Barn Swallow, Cliff Swallow, Eastern Phoebe) may build their nests on structures such as bridges, ledges or gutters.

- One method frequently used to minimize the risk of destroying bird nests consists of avoiding certain activities, such as clearing, during the regional nesting period for migratory birds.

- The risk of impacting active nests or birds caring for pre-fledged chicks, discovered during project activities outside the regional nesting period, can be minimized by measures such as the establishment of vegetated buffer zones around nests, and minimization of activities in the immediate area until nesting is complete and chicks have naturally migrated from the area. It is incumbent on the proponent to identify the best approach, based on the circumstances, to complying with the MBCA.

  Further information can be found at https://www.canada.ca/en/environment-climate-change/services/avoiding-harm-migratory-birds.html

The proponent should also be reminded that the prohibitions under the *Species at Risk Act* (SARA) are now in force. The complete text of SARA, including prohibitions, is available at www.sararegistry.gc.ca .

Please don't hesitate to contact me should you have any questions regarding our comments.

Sincerely,

Rachel
Rachel Gautreau
Coordinator, Environmental Assessment / Canadian Wildlife Service
Environment and Climate Change Canada / Government of Canada
rachel.gautreau@canada.ca

Coordinatrice, Évaluations environnementales / Service canadien de la faune
Environnement et Changement climatique Canada / Gouvernement du Canada
rachel.gautreau@canada.ca

Town Point Consulting Inc.

March 3, 2021

NS Department of Fisheries and Aquaculture
1575 Lake Road
Shelburne, NS
B0T 1W0

Attention; Nathaniel Fiendel, Lewis Clancey

AQ1442,1443,1444

Please see below our reply to the Network Agency Review received from Canadian Wildlife Service.

Canadian Wildlife Service

1) We understand the sensitivity of Plover habitat on Dunn's Beach. The scientists we have consulted on this matter stated that Plovers use the ocean facing beach and are unlikely to use the harbour side of the beach. They also stated that a 200m buffer is reasonable and has been demonstrated to work in other situations involving more disruptive operations adjacent to Plover habitat. Is it possible to revise the requirement for a 300m setback and agree on a 250m buffer instead given that this buffer would be from the harbour side of the beach and plovers use the ocean side which is further away?

2) Some portions of Dunn's Beach are wooded with a boulder harbour shore – not Plover habitat. Is it necessary to apply the setback requirements to these portions of the Dunn's Beach peninsula?

3) The BOBR growth units to be used on this farm are far less attractive to seabirds due to inherent instability and lack of buoyancy. We understand the issues associated with birds roosting on floating cages and believe we have very effectively mitigated this risk. During 17 months of trials no cormorants roosts on BOBR but were commonly seen on Oyster Gro cages nearby.

4) What species of Gull is known to nest on Gooseberry Island. Where is the data associated with this expectation? What was the source of this data and when was the data obtained? What population of Gulls is known to nest on this island? The principle view from our house is Gooseberry Island and we frequently boat nearby. Never have we observed gulls nesting on or frequenting near this Island.

5) Is there a way to accommodate the possibility of gull nesting some way without eliminating this site from our farm plan?

Dr. Tony Miller professor emeritus from St FX came to our site on Wednesday Feb. 24, 2021. Tony is a biology prof who studied Antigonish Harbour extensively for many decades and frequently canoes there to study birds.
Tony came to my property on Wednesday for the afternoon and we spent 31/2 hours discussing our farm plan and its potential impact on birds.

Points of note from this discussion are as follows;

1) He knows of only one gull species that nests on Gooseberry Island. It is the Black Backed Gull. He said annually between 10 and 20 of these birds nest there. He has often approached them by canoe and says they do not leave their nest until you approach on foot along the shore and at a close distance. He also stated the nests are located on the western shore of the island...the side closest to the harbour channel and distant from our proposed lease site. He does not believe our farming operation would disturb these nesting birds. He is very curious to know what other gull is claimed to nest there because he has no knowledge of a second species connected to the island.

2) Regarding Plovers on Dunn's Beach, Dr. Miller agrees they use the ocean side not the harbour side of the beach and he did not see any problem with the proposed operation relative to plovers. He identified the area used by Turns which is near the western tip of the beach, also sufficiently away from the lease sites.

3) Regarding the Captain's Island site, Dr. Miller had no concerns. He brought up geese repeatedly but always pointed out that they use the area between Captain's Island and Mahoney's Beach. Dr. Miller said several times that he sees no reason our plan would present a problem for birds that use the estuary.

As discussed, we would welcome a meeting with CWS to discuss the concerns they have expressed and to explore ways to mitigate these concerns.

Best regards,
Ernie Porter, P.Eng., President

Town Point Consulting Inc.

Applicant Submitted Videos of Gooseberry Island

See Videos #1843 (May 30, 2021) and #1925 (Dec.20, 2020) Gooseberry Island on the stick drive provided.

From: Clancey, Lewis
Sent: March 9, 2021 11:34 AM
To: rachel.gautreau@canada.ca
Cc: Nathaniel Feindel <nathaniel.feindel@novascotia.ca>
Subject: Aquaculture Lease Application #'s 1442, 1443, 1444, Town Point Consulting Inc.

Hi Rachel,

We are wondering if you would be available to meet with the applicant regarding the lease applications for oyster culture in Antigonish Harbour. We are targeting a meeting time of sometime next week, 15-20 February, 2021.

The applicant has received your network agency review comments and would like to have an opportunity to discuss the identified issues your department.

Would you, or a representative from your department, be available to meet with NSDFA and the applicant sometime next week?

Thank you,

Lew

Lewis Clancey
Aquaculture Development Advisor
NS Dept. of Fisheries and Aquaculture
1800 Argyle St.
WTCC 6th Floor, Suite 604,
Halifax, NS
B3J2R5
902 956 3839
lewis.clancey@novascotia.ca

From: Clancey, Lewis
Sent: May 17, 2021 10:05 AM
To: rachel.gautreau@canada.ca
Cc: Spencer, Amanda L <Amanda.Spencer@novascotia.ca>
Subject: Antigonish Oyster Lease Applications 1442, 1443, 1444

Hi Rachel,

Nathaniel asked me to send a note to see if we could have a follow-up discussion about the Antigonish oyster lease applications.

The applicant would like to discuss these further but, at this time, we feel that discussions between our departments would be more appropriate.

Could you check with your personnel to see if they would be able to have another discussion and suggest a few dates for this to occur?

Thanks,

Lew

Lewis Clancey
Aquaculture Development Advisor
NS Dept. of Fisheries and Aquaculture
1800 Argyle St.
WTCC 6th Floor, Suite 604,
Halifax, NS
B3J2R5

902 956 3839
lewis.clancey@novascotia.ca

From: Clancey, Lewis
Sent: May 27, 2021 3:18 PM
To: rachel.gautreau@canada.ca
Cc: joshua.mailhiot@canada.ca
Subject: FW: Antigonish Oyster Lease Applications 1442, 1443, 1444

Hi Rachel,

I'm just sending a friendly reminder for my previous email, below. Would you, and your staff, be able to suggest a few meetings dates that are convenient for you?

Thanks,

Lew

From: Clancey, Lewis <Lewis.Clancey@novascotia.ca>
Sent: June 21, 2021 10:50 AM
To: rachel.gautreau@canada.ca
Cc: joshua.mailhiot@canada.ca; Feindel, Nathaniel J <Nathaniel.Feindel@novascotia.ca>
Subject: RE: Antigonish Oyster Lease Applications 1442, 1443, 1444

Hi Rachelle,

Any thoughts on proposed dates for a meeting with out group regarding the Antigonish leases?

Thanks,

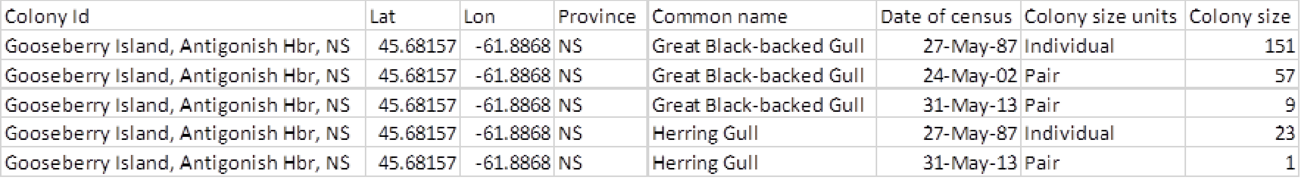
Lew

** EXTERNAL EMAIL / COURRIEL EXTERNE **
Exercise caution when opening attachments or clicking on links / Faites preuve de prudence si vous ouvrez une pièce jointe ou cliquez sur un lien

Hello Nathaniel,

Environment and Climate Change Canada's Canadian Wildlife Service has completed an additional review of the application based on the new information presented. The additional information did not result in any changes to the advice provided on December 7, 2020 regarding buffers around Gooseberry Island and Dunn's Beach.

It appears that the proponent is questioning the use of Gooseberry Island by colonial nesters. Censuses of colonial waterbird nesting sites are conducted periodically, and our data on colonial nester use of this island is as follows:



Colony Id	Lat	Lon	Province	Common name	Date of census	Colony size units	Colony size
Gooseberry Island, Antigonish Hbr, NS	45.68157	-61.8868	NS	Great Black-backed Gull	27-May-87	Individual	151
Gooseberry Island, Antigonish Hbr, NS	45.68157	-61.8868	NS	Great Black-backed Gull	24-May-02	Pair	57
Gooseberry Island, Antigonish Hbr, NS	45.68157	-61.8868	NS	Great Black-backed Gull	31-May-13	Pair	9
Gooseberry Island, Antigonish Hbr, NS	45.68157	-61.8868	NS	Herring Gull	27-May-87	Individual	23
Gooseberry Island, Antigonish Hbr, NS	45.68157	-61.8868	NS	Herring Gull	31-May-13	Pair	1

It is not unusual for colonial waterbirds on smaller islands to not be present every year, and the sizes of smaller colonies are often variable.

In order to avoid disturbance to sensitive species, CWS recommends a two-pronged approach: establishment of buffer zones around sensitive features where no aquaculture leases are sited, and measures taken by aquaculture operators to avoid adverse effects on sensitive species as well as other migratory birds.

During the review, concern was raised regarding the proponent accessing Gooseberry Island despite our comment that there should be no access to the island, including the intertidal zone, during spring and summer by project staff and/or equipment. While the proponent did not find gulls on the island, their presence would have caused disturbance had colonial waterbirds been present in 2021. As a reminder, migratory birds, including gulls, are protected under the *Migratory Birds Convention Act* and associated regulations.

Canadian Wildlife Service can be available for a meeting if you have further questions. Please contact me if you wish to coordinate a meeting.

Sincerely,

Rachel

APPENDIX F: NS DEPARTMENT OF ENVIRONMENT (NOW DEPARTMENT OF ENVIRONMENT AND CLIMATE CHANGE)

From: Greenwood, Megan N
Sent: October 7, 2020 2:26 PM
To: chris.mills@dfo-mpo.gc.ca; Braid, Leanna <Leanna.Braid@dfo-mpo.gc.ca>; Smith, Angela (CFIA/ACIA) <angela.smith@canada.ca>; david.macarthur@ec.gc.ca; Birch, Angela <Angela.Birch@novascotia.ca>; Kittilsen, Michael A <Michael.Kittilsen@novascotia.ca>; Cottreau-Robins, Catherine M <Catherine.Cottreau-Robins@novascotia.ca>; Blackburn, Lori M <Lori.Blackburn@novascotia.ca>; Boudreau, Louise O <Louise.Boudreau@novascotia.ca>; Gautreau, Rachel (EC) <rachel.gautreau@canada.ca>
Cc: Feindel, Nathaniel J <Nathaniel.Feindel@novascotia.ca>; Clancey, Lewis <Lewis.Clancey@novascotia.ca>; Ceschiutti, Robert <Robert.Ceschiutti@novascotia.ca>; King, Matthew S <Matthew.King@novascotia.ca>; Hancock, Bruce H <Bruce.Hancock@novascotia.ca>
Subject: AQ#1442, 1443, 1444 - New Application

Attention: Network Review Agencies

Please see the attached memo for three (3) Aquaculture Application Nos. 1442, 1443, 1444 in Antigonish Harbour, Antigonish County.

Please respond with your feedback by **December 7, 2020.**

The attached zipped shapefile requires specific geospatial software to open and is intended for convenient use by GIS professionals only.

Megan Greenwood

Licensing Coordinator
Nova Scotia Department of Fisheries and Aquaculture
1575 Lake Road
Shelburne, NS
B0T 1W0
Phone 902-875-7443
Fax 902-875-7429

Network Agency Review of an Aquaculture Application

Agency	**DOE**
Division (if applicable)	**Enforcement and Compliance**
Date	**October 29, 2020**
File No.	**1444**
Type of application	**New Marine Shellfish**
Information Provided	**Peter Taylor**

Please provide comments, concerns, recommendations, or requirements on the above stated application for a marine aquaculture licence. Please include the criterion /criteria within your jurisdiction or mandate that your feedback is based upon. Similarly, if additional information is required to make a determination, please include the criterion /criteria within your jurisdiction or mandate that your request is based upon.

☐ No concerns regarding the proposed development
☒ Concerns with development are expressed below
☐ Request modifications to the proposed development (described below)
☐ Required or recommended conditions (described below)
☐ Request additional information (described below)
☐ Request meeting with applicant and NSDFA (described below)
☐ No comments on the application

Comments, concerns, recommendations, and/or required conditions including the criterion /criteria within your jurisdiction or mandate that your feedback is based upon. (Attach comments if preferred, or add additional pages, as required.):

Sites could/will effect:
Other commercial oyster fishers-- oysters are openly collected outside the 1 AQ site # 1385, throughout the harbour and being flushed in approved areas before sale or to another AQ site
Commercial and recreational ell harvesters
Float plane operation
All types of recreational boating
Recreational fishing
Commercial lobster fishers travel?
these activities should be addressed if not already.

Public Notice and Disclosure

As part of the process for deciding on an application, it may be necessary for the Nova Scotia Department of Fisheries and Aquaculture (“Fisheries and Aquaculture”) to disclose the collected network review information to the applicant and other government bodies, including, if applicable, the Nova Scotia Aquaculture Review Board for use at an adjudicative hearing relating to the application in question.

In accordance with departmental policy, which seeks to promote public involvement in the process for deciding on aquaculture applications, Fisheries and Aquaculture will disclose aquaculture application information, including network review information, on the departmental website.

Privacy Statement

The network review information collected as part of an aquaculture application will only be used or disclosed by Fisheries and Aquaculture for the purpose of deciding on the application.

All application information collected is subject to the Freedom of Information and Protection of Privacy Act (“FOIPOP”) and will only be used or disclosed in accordance with FOIPOP.

From: MacKay, Troy
To: Greenwood, Megan N
Subject: Aquaculture license and lease applications #1442,1443 and 1444
Date: January 7, 2021 4:29:24 PM
Attachments: Network Agency Review-AQ#1444.docx
Network Agency Review-AQ#1443.docx
Network Agency Review-AQ#1442.docx
image002.png

Hi Meghan,

Just reviewed the network agency review that one of my officers had filled out, regarding aquaculture license and lease applications #1442,1443 and 1444 and his comments expressed within these documents. The comments should have been that we in the Conservation Officer Service have no compliance issues to date in these areas. He misunderstood the question and the other comments that were expressed, were a personal opinion and outside this Departments mandate.

NOVA SCOTIA
Environment

Troy MacKay
Regional Manager,
Conservation Officer Service
Inspection Compliance & Enforcement Division
300 Mountain Road
Coxheath, NS B1L 1A9
☎ (902) 563-3370
📠 (902) 567-2535
✉ troy.mackay@novascotia.ca
Toll Free: 1-800-565-2224 to Report Violations
Or Email: reportapoacher@novascotia.ca

APPENDIX G: NS COMMUNITIES, CULTURE AND HERITAGE (NOW DEPARTMENT OF COMMUNITIES, CULTURE, TOURISM AND HERITAGE)

From: Greenwood, Megan N
Sent: October 7, 2020 2:26 PM
To: chris.mills@dfo-mpo.gc.ca; Braid, Leanna <Leanna.Braid@dfo-mpo.gc.ca>; Smith, Angela (CFIA/ACIA) <angela.smith@canada.ca>; david.macarthur@ec.gc.ca; Birch, Angela <Angela.Birch@novascotia.ca>; Kittilsen, Michael A <Michael.Kittilsen@novascotia.ca>; Cottreau-Robins, Catherine M <Catherine.Cottreau-Robins@novascotia.ca>; Blackburn, Lori M <Lori.Blackburn@novascotia.ca>; Boudreau, Louise O <Louise.Boudreau@novascotia.ca>; Gautreau, Rachel (EC) <rachel.gautreau@canada.ca>
Cc: Feindel, Nathaniel J <Nathaniel.Feindel@novascotia.ca>; Clancey, Lewis <Lewis.Clancey@novascotia.ca>; Ceschiutti, Robert <Robert.Ceschiutti@novascotia.ca>; King, Matthew S <Matthew.King@novascotia.ca>; Hancock, Bruce H <Bruce.Hancock@novascotia.ca>
Subject: AQ#1442, 1443, 1444 - New Application

Attention: Network Review Agencies

Please see the attached memo for three (3) Aquaculture Application Nos. 1442, 1443, 1444 in Antigonish Harbour, Antigonish County.

Please respond with your feedback by **December 7, 2020.**

The attached zipped shapefile requires specific geospatial software to open and is intended for convenient use by GIS professionals only.

Megan Greenwood

Licensing Coordinator
Nova Scotia Department of Fisheries and Aquaculture
1575 Lake Road
Shelburne, NS
B0T 1W0
Phone 902-875-7443
Fax 902-875-7429

Network Agency Review of an Aquaculture Application

Agency	**CCH**
Division (if applicable)	**Special Places**
Date	**Nov. 25, 2020**
File No.	**1444**
Type of application	**New Marine Shellfish**
Information Provided	**archaeology**

Please provide comments, concerns, recommendations, or requirements on the above stated application for a marine aquaculture licence. Please include the criterion /criteria within your jurisdiction or mandate that your feedback is based upon. Similarly, if additional information is required to make a determination, please include the criterion /criteria within your jurisdiction or mandate that your request is based upon.

☐ No concerns regarding the proposed development
☒ Concerns with development are expressed below
☐ Request modifications to the proposed development (described below)
☐ Required or recommended conditions (described below)
☐ Request additional information (described below)
☐ Request meeting with applicant and NSDFA (described below)
☐ No comments on the application

Comments, concerns, recommendations, and/or required conditions including the criterion /criteria within your jurisdiction or mandate that your feedback is based upon. (Attach comments if preferred, or add additional pages, as required.):

This is an area of elevated archaeological potential. Given this is a suspended culture set up, immediate archaeology concerns are minimal. However, please communicate to the license operator, that if at anytime artifacts are encountered during operations, the Coordinator of Special Places, John Cormier should be contacted immediately.

Public Notice and Disclosure

As part of the process for deciding on an application, it may be necessary for the Nova Scotia Department of Fisheries and Aquaculture ("Fisheries and Aquaculture") to disclose the collected network review information to the applicant and other government bodies, including, if applicable, the Nova Scotia Aquaculture Review Board for use at an adjudicative hearing relating to the application in question.

In accordance with departmental policy, which seeks to promote public involvement in the process for deciding on aquaculture applications, Fisheries and Aquaculture will disclose aquaculture application information, including network review information, on the departmental website.

Privacy Statement

The network review information collected as part of an aquaculture application will only be used or disclosed by Fisheries and Aquaculture for the purpose of deciding on the application.

All application information collected is subject to the Freedom of Information and Protection of Privacy Act ("FOIPOP") and will only be used or disclosed in accordance with FOIPOP.

From: Ceschiutti, Robert
To: Shore, Christopher; Lewis, Beth J
Cc: Winfield, Lynn; Buchan, Carla M
Subject: Updates to Aquaculture applications AQ#1442 1443 1444
Date: August 31, 2022 1:18:50 PM
Attachments: 2020.11.25-CCH-Catherine Robbins-Cottreau-AQ#1442.pdf
2020-11-25 CCH-Catherine Robbins-Cottreau-AQ#1443-2020.11.25.pdf
2020-11-25 CCH-Catherine Robbins-Cottreau-Response-AQ#1444-2020.11.25.pdf

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This email is in regards to three new marine shellfish Aquaculture applications, file nos. AQ#1442, 1443 and 1444 (Town Point Consulting Ltd.). I wish to inform you that the Kwilmu'kw Maw'klusuaqn Negotiation Office (KMKNO) had sent correspondence to our department (NSDFA) regarding the above applications on July 13, 2022. In the letter, the KMKNO recommended a full Archaeological Resource Impact Assessment (ARIA) be undertaken. The KMKNO did not provide an explanation or specificity for how the proposed suspended culture activities could impact rights related to underwater archaeology in the project area.

Could you please confirm if your previous response, dated November 25, 2020 remains valid?

Regards,
Robert Ceschiutti
Manager, Licensing and Leasing
NS Department of Fisheries and Aquaculture
1575A Lake Road
Sandy Point, Nova Scotia
B0T 1W0
Phone: 902-875-7430
Robert.Ceschiutti@novascotia.ca

From: Ceschiutti, Robert
To: Shore, Christopher; Lewis, Beth J; Cottreau-Robins, Catherine M
Cc: Winfield, Lynn; Buchan, Carla M
Subject: RE: Updates to Aquaculture applications AQ#1442 1443 1444
Date: September 12, 2022 10:45:02 AM

I am following up of the previous email sent to you on August 31, 2022 in regards to Aquaculture applications AQ#1442, 1443 and 1444 (Town Point Consulting Ltd.).

Can you please respond to my initial request by the end of today?

Regards,
Robert Ceschiutti
Manager, Licensing and Leasing
NS Department of Fisheries and Aquaculture
1575A Lake Road
Sandy Point, Nova Scotia
B0T 1W0
Phone: 902-875-7430
Robert.Ceschiutti@novascotia.ca

From: Cottreau-Robins, Catherine M
To: Ceschiutti, Robert; Shore, Christopher; Lewis, Beth J
Cc: Winfield, Lynn; Buchan, Carla M
Subject: RE: Updates to Aquaculture applications AQ#1442 1443 1444
Date: September 12, 2022 11:08:36 AM

Hi Robert,

I will look this up again and get back to you. I have been on vacation leave and then got Covid. I hope to return to work tomorrow.
I will say that the archaeological concerns by the KMKNO are not unexpected.

Yours,

Katie Cottreau-Robins

From: Ceschiutti, Robert
To: Winfield, Lynn; Feindel, Nathaniel J; Clancey, Lewis; Buchan, Carla M
Subject: FW: Updates to Aquaculture applications AQ#1442 1443 1444
Date: September 13, 2022 8:07:11 AM

Hi team, I received a response from CCTH via OLA below that can be added to the record for AQ#1442 1443 and 1444.

Regards,
Robert Ceschiutti
Manager, Licensing and Leasing
NS Department of Fisheries and Aquaculture
1575A Lake Road
Sandy Point, Nova Scotia
B0T 1W0
Phone: 902-875-7430
Robert.Ceschiutti@novascotia.ca

From: Rillie, Claire Z <Claire.Rillie@novascotia.ca>
Sent: September 12, 2022 3:47 PM
To: Ceschiutti, Robert <Robert.Ceschiutti@novascotia.ca>
Subject: FW: Updates to Aquaculture applications AQ#1442 1443 1444

Hi Robert,

Please see CCTH's responses to your questions about the above-noted files, below. Happy to discuss if needed.

Thanks,
Claire

From: Lewis, Beth J <Beth.Lewis@novascotia.ca>
Sent: September 12, 2022 1:55 PM
To: Cottreau-Robins, Catherine M <Catherine.Cottreau-Robins@novascotia.ca>; Rillie, Claire Z <Claire.Rillie@novascotia.ca>; Cormier, John Kenneth <John.Cormier@novascotia.ca>
Subject: RE: Updates to Aquaculture applications AQ#1442 1443 1444

Thank you Katie and John for the fast turn around. Katie, I hope you're feeling okay.

Claire, would you like to follow up with Robert or is it easier for me to? I'm fine either way, just let me know.

B

From: Cottreau-Robins, Catherine M <Catherine.Cottreau-Robins@novascotia.ca>

Sent: September 12, 2022 11:27 AM
To: Rillie, Claire Z <Claire.Rillie@novascotia.ca>; Lewis, Beth J <Beth.Lewis@novascotia.ca>; Cormier, John Kenneth <John.Cormier@novascotia.ca>
Subject: RE: Updates to Aquaculture applications AQ#1442 1443 1444

Hi Folks,

A brief letter. Please note that when CCTH responds with "this is an area of elevated archaeological potential," we, of course, include all potential archaeological resources – indigenous (historic and pre-contact), historic, and marine/shipwreck.

If Aquaculture wants to support a full ARIA that is fine but again, suspended culture operations means suspended equipment except for anchors that I have been assuured , do not drag across the ocean floor and are checked for such impacts regularly. Anchors often sit upon considerable ocean sediment buildup.

Hope this helps.

I am off on Mondays as you know, but back to work tomorrow.

Katie

From: Rillie, Claire Z <Claire.Rillie@novascotia.ca>
Sent: September 12, 2022 11:15 AM
To: Cottreau-Robins, Catherine M <Catherine.Cottreau-Robins@novascotia.ca>; Lewis, Beth J <Beth.Lewis@novascotia.ca>; Cormier, John Kenneth <John.Cormier@novascotia.ca>
Subject: RE: Updates to Aquaculture applications AQ#1442 1443 1444

Hi Folks,

Apologies for the confusion on this file. I'm attaching KMKNO's letter and I'd be happy to answer any more questions you may have.

Thanks,
Claire

From: Cottreau-Robins, Catherine M <Catherine.Cottreau-Robins@novascotia.ca>
Sent: September 12, 2022 11:10 AM
To: Lewis, Beth J <Beth.Lewis@novascotia.ca>; Cormier, John Kenneth <John.Cormier@novascotia.ca>; Rillie, Claire Z <Claire.Rillie@novascotia.ca>
Subject: RE: Updates to Aquaculture applications AQ#1442 1443 1444

Looking at this now Beth. katie

From: Lewis, Beth J <Beth.Lewis@novascotia.ca>
Sent: September 12, 2022 10:52 AM
To: Cottreau-Robins, Catherine M <Catherine.Cottreau-Robins@novascotia.ca>; Cormier, John Kenneth <John.Cormier@novascotia.ca>; Rillie, Claire Z <Claire.Rillie@novascotia.ca>
Subject: FW: Updates to Aquaculture applications AQ#1442 1443 1444

Hi John and Katie,

Just noticing neither of you were not copied on the original. I'm wondering if you are able to meet Robert's requested response time by the end of the day to confirm our original assessment/response from November 2020 (attached). If not, please let me know when we can provide a response and I'll let Robert know our timeline.

Claire, can you provide copy of the letter that KMKNO sent regarding this project so that Katie and John can reference?

Thank you,
Beth

From: Ceschiutti, Robert
To: Cottreau-Robins, Catherine M
Cc: Buchan, Carla M; Clancey, Lewis; Winfield, Lynn
Subject: FW: Updates to Aquaculture applications AQ#1442 1443 1444
Date: September 13, 2022 12:03:01 PM
Importance: High

Good afternoon Catherine, because these files are related to adjudicative applications, it's important that I have a direct response from CCTH to my email request. Can you please respond directly to me that your previous response, dated November 25, 2020 remains valid?

Regards,
Robert Ceschiutti
Manager, Licensing and Leasing
NS Department of Fisheries and Aquaculture
1575A Lake Road
Sandy Point, Nova Scotia
B0T 1W0
Phone: 902-875-7430
Robert.Ceschiutti@novascotia.ca

From: Cottreau-Robins, Catherine M
To: Ceschiutti, Robert
Cc: Buchan, Carla M; Clancey, Lewis; Winfield, Lynn
Subject: Re: Updates to Aquaculture applications AQ#1442 1443 1444
Date: September 22, 2022 8:45:08 AM

Hi Robert,

I believe I responded to this but will confirm. I am in Membertou today but should be able to address tomorrow.

Yours,

Katie

From: Ceschiutti, Robert
To: Cottreau-Robins, Catherine M
Cc: Buchan, Carla M; Clancey, Lewis; Winfield, Lynn
Subject: RE: Updates to Aquaculture applications AQ#1442 1443 1444
Date: September 27, 2022 11:01:00 AM

Hi Katie, to clarify you did send a response to Claire Rillie from the Nova Scotia Office of L'Nu Affairs on September 12, 2022 but I require a separate email sent directly to myself (representing the NSDFA) as opposed to another Department (OLA). Please send the response ASAP, thank you.

Regards,
Robert Ceschiutti
Manager, Licensing and Leasing
NS Department of Fisheries and Aquaculture
1575A Lake Road
Sandy Point, Nova Scotia
B0T 1W0
Phone: 902-875-7430
Robert.Ceschiutti@novascotia.ca

From: Cottreau-Robins, Catherine M <Catherine.Cottreau-Robins@novascotia.ca>
Sent: September 27, 2022 12:33 PM
To: Ceschiutti, Robert <Robert.Ceschiutti@novascotia.ca>; Cormier, John Kenneth <John.Cormier@novascotia.ca>; Lewis, Beth J <Beth.Lewis@novascotia.ca>
Cc: Buchan, Carla M <Carla.Buchan@novascotia.ca>; Clancey, Lewis <Lewis.Clancey@novascotia.ca>; Winfield, Lynn <Lynn.Winfield@novascotia.ca>; Rillie, Claire Z <Claire.Rillie@novascotia.ca>
Subject: RE: Updates to Aquaculture applications AQ#1442 1443 1444

Hi Robert,

I have been without power but back up and looked at all this again.

Unless the aquaculture operation type has changed from what I reviewed in Nov 2020, my assessment is the same. These are suspended culture operations that have low impact given the anchors are stationary. However, because this is an area of general elevated archaeological potential, communication on any artifact findings to the Special Places Office is requested so we can follow up.

Yours,

Katie Cottreau-Robins

From: Ceschiutti, Robert
To: Cottreau-Robins, Catherine M; Cormier, John Kenneth; Lewis, Beth J
Cc: Buchan, Carla M; Clancey, Lewis; Winfield, Lynn; Rillie, Claire Z
Subject: RE: Updates to Aquaculture applications AQ#1442 1443 1444
Date: September 28, 2022 8:10:13 AM

Perfect, thanks!

Regards,
Robert Ceschiutti
Manager, Licensing and Leasing
NS Department of Fisheries and Aquaculture
1575A Lake Road
Sandy Point, Nova Scotia
B0T 1W0
Phone: 902-875-7430
Robert.Ceschiutti@novascotia.ca

From: Feindel, Nathaniel J
To: Winfield, Lynn; Clancey, Lewis
Subject: FW: Town Point Consulting Inc.
Date: November 21, 2022 10:46:38 AM

Hey Lynn and Lew,

I had a conversation with Catherine last week as per my email below.

Lew, please included this in the report on consultation under CCTH's section.

Thanks, Nathaniel

From: Feindel, Nathaniel J
Sent: November 21, 2022 10:45 AM
To: Cottreau-Robins, Catherine M <Catherine.Cottreau-Robins@novascotia.ca>
Subject: Town Point Consulting Inc.

Hello Catherine,

Thanks for the conversation last Thursday (Nov 17th), regarding some of the similarities and differences in the standard suspended oyster growing infrastructure and the technology proposed to be utilized for suspended culture by Town Point Consulting Inc., in Antigonish Harbour. We felt it was important to clarify culture methodologies to ensure the advice provided by CCTH on this file was appropriate.

As you outlined in during our conversation and in the advice provided by CCTH to date, the proposed area has high archaeological potential. The next stage is for the application to move it to the Aquaculture Review Board for a decision on the applications. We will keep you informed on this and if it is approved, we will work with the operator to ensure the appropriate reporting mechanisms and procedures are in place, should farming commence and if archeological artifacts are encountered.

Thanks, Nathaniel

Nathaniel Feindel
Aquaculture Development and Marine Plants Harvesting- Manager
N.S. Dept. Fisheries & Aquaculture
1575A Lake Rd., Sandy Point, N.S., B0T1W0
T: (902) 875-7450
F: (902) 875-7429
E: Nathaniel.Feindel@novascotia.ca

From: Feindel, Nathaniel J
To: Clancey, Lewis
Cc: Winfield, Lynn
Subject: FW: Town Point Consulting Inc.
Date: November 24, 2022 10:26:34 AM

Hey Lew,

Please see the correspondence with CCTH.

Nathaniel

From: Cottreau-Robins, Catherine M <Catherine.Cottreau-Robins@novascotia.ca>
Sent: November 22, 2022 9:18 AM
To: Feindel, Nathaniel J <Nathaniel.Feindel@novascotia.ca>
Cc: Cormier, John Kenneth <John.Cormier@novascotia.ca>
Subject: RE: Town Point Consulting Inc.

THank you Nathaniel. katie

From: Feindel, Nathaniel J <Nathaniel.Feindel@novascotia.ca>
Sent: November 21, 2022 10:45 AM
To: Cottreau-Robins, Catherine M <Catherine.Cottreau-Robins@novascotia.ca>
Subject: Town Point Consulting Inc.

Hello Catherine,

Thanks for the conversation last Thursday (Nov 17th), regarding some of the similarities and differences in the standard suspended oyster growing infrastructure and the technology proposed to be utilized for suspended culture by Town Point Consulting Inc., in Antigonish Harbour. We felt it was important to clarify culture methodologies to ensure the advice provided by CCTH on this file was appropriate.

As you outlined in during our conversation and in the advice provided by CCTH to date, the proposed area has high archaeological potential. The next stage is for the application to move it to the Aquaculture Review Board for a decision on the applications. We will keep you informed on this and if it is approved, we will work with the operator to ensure the appropriate reporting mechanisms and procedures are in place, should farming commence and if archeological artifacts are encountered.

Thanks, Nathaniel

Nathaniel Feindel
Aquaculture Development and Marine Plants Harvesting- Manager
N.S. Dept. Fisheries & Aquaculture
1575A Lake Rd., Sandy Point, N.S., B0T1W0

T: (902) 875-7450
F: (902) 875-7429
E: Nathaniel.Feindel@novascotia.ca

APPENDIX H: NS DEPARTMENT OF AGRICULTURE

From: Greenwood, Megan N
Sent: October 7, 2020 2:26 PM
To: chris.mills@dfo-mpo.gc.ca; Braid, Leanna <Leanna.Braid@dfo-mpo.gc.ca>; Smith, Angela (CFIA/ACIA) <angela.smith@canada.ca>; david.macarthur@ec.gc.ca; Birch, Angela <Angela.Birch@novascotia.ca>; Kittilsen, Michael A <Michael.Kittilsen@novascotia.ca>; Cottreau-Robins, Catherine M <Catherine.Cottreau-Robins@novascotia.ca>; Blackburn, Lori M <Lori.Blackburn@novascotia.ca>; Boudreau, Louise O <Louise.Boudreau@novascotia.ca>; Gautreau, Rachel (EC) <rachel.gautreau@canada.ca>
Cc: Feindel, Nathaniel J <Nathaniel.Feindel@novascotia.ca>; Clancey, Lewis <Lewis.Clancey@novascotia.ca>; Ceschiutti, Robert <Robert.Ceschiutti@novascotia.ca>; King, Matthew S <Matthew.King@novascotia.ca>; Hancock, Bruce H <Bruce.Hancock@novascotia.ca>
Subject: AQ#1442, 1443, 1444 - New Application

Attention: Network Review Agencies

Please see the attached memo for three (3) Aquaculture Application Nos. 1442, 1443, 1444 in Antigonish Harbour, Antigonish County.

Please respond with your feedback by **December 7, 2020.**

The attached zipped shapefile requires specific geospatial software to open and is intended for convenient use by GIS professionals only.

Megan Greenwood

Licensing Coordinator
Nova Scotia Department of Fisheries and Aquaculture
1575 Lake Road
Shelburne, NS
B0T 1W0
Phone 902-875-7443
Fax 902-875-7429

Network Agency Review of an Aquaculture Application

Agency	**NS Department of Agriculture**
Division (if applicable)	**Animal and Crop Services**
Date	**November 10, 2020**
File No.	**1444**
Type of application	**New Marine Shellfish**
Information Provided	

Please provide comments, concerns, recommendations, or requirements on the above stated application for a marine aquaculture licence. Please include the criterion /criteria within your jurisdiction or mandate that your feedback is based upon. Similarly, if additional information is required to make a determination, please include the criterion /criteria within your jurisdiction or mandate that your request is based upon.

☒ No concerns regarding the proposed development
☐ Concerns with development are expressed below
☐ Request modifications to the proposed development (described below)
☐ Required or recommended conditions (described below)
☐ Request additional information (described below)
☐ Request meeting with applicant and NSDFA (described below)
☐ No comments on the application

Comments, concerns, recommendations, and/or required conditions including the criterion /criteria within your jurisdiction or mandate that your feedback is based upon. (Attach comments if preferred, or add additional pages, as required.):

There are no concerns from NSDA currently, agriculture operations near the purposed site 1444 are light in intensity. The intensity of the agricultural land use may change over time and become more intense. If the agricultural operations become more intense there is the potential for agricultural runoff to occur causing a negative impact on the purposed Aquiculture operation.

Public Notice and Disclosure

As part of the process for deciding on an application, it may be necessary for the Nova Scotia Department of Fisheries and Aquaculture (“Fisheries and Aquaculture”) to disclose the collected network review information to the applicant and other government bodies, including, if applicable, the Nova Scotia Aquaculture Review Board for use at an adjudicative hearing relating to the application in question.

In accordance with departmental policy, which seeks to promote public involvement in the process for deciding on aquaculture applications, Fisheries and Aquaculture will disclose aquaculture application information, including network review information, on the departmental website.

Privacy Statement

The network review information collected as part of an aquaculture application will only be used or disclosed by Fisheries and Aquaculture for the purpose of deciding on the application.

All application information collected is subject to the Freedom of Information and Protection of Privacy Act (“FOIPOP”) and will only be used or disclosed in accordance with FOIPOP.

APPENDIX I: NS DEPARTMENT OF LANDS AND FORESTRY (NOW DEPARTMENT OF NATURAL RESOURCES AND RENEWABLES)

From: Greenwood, Megan N
Sent: October 7, 2020 2:26 PM
To: chris.mills@dfo-mpo.gc.ca; Braid, Leanna <Leanna.Braid@dfo-mpo.gc.ca>; Smith, Angela (CFIA/ACIA) <angela.smith@canada.ca>; david.macarthur@ec.gc.ca; Birch, Angela <Angela.Birch@novascotia.ca>; Kittilsen, Michael A <Michael.Kittilsen@novascotia.ca>; Cottreau-Robins, Catherine M <Catherine.Cottreau-Robins@novascotia.ca>; Blackburn, Lori M <Lori.Blackburn@novascotia.ca>; Boudreau, Louise O <Louise.Boudreau@novascotia.ca>; Gautreau, Rachel (EC) <rachel.gautreau@canada.ca>
Cc: Feindel, Nathaniel J <Nathaniel.Feindel@novascotia.ca>; Clancey, Lewis <Lewis.Clancey@novascotia.ca>; Ceschiutti, Robert <Robert.Ceschiutti@novascotia.ca>; King, Matthew S <Matthew.King@novascotia.ca>; Hancock, Bruce H <Bruce.Hancock@novascotia.ca>
Subject: AQ#1442, 1443, 1444 - New Application

Attention: Network Review Agencies

Please see the attached memo for three (3) Aquaculture Application Nos. 1442, 1443, 1444 in Antigonish Harbour, Antigonish County.

Please respond with your feedback by **December 7, 2020.**

The attached zipped shapefile requires specific geospatial software to open and is intended for convenient use by GIS professionals only.

Megan Greenwood

Licensing Coordinator
Nova Scotia Department of Fisheries and Aquaculture
1575 Lake Road
Shelburne, NS
B0T 1W0
Phone 902-875-7443
Fax 902-875-7429

From: Greenwood, Megan N <Megan.Greenwood@novascotia.ca>
Sent: November 24, 2020 10:49 AM
To: chris.mills@dfo-mpo.gc.ca; Braid, Leanna <Leanna.Braid@dfo-mpo.gc.ca>; Smith, Angela (CFIA/ACIA) <angela.smith@canada.ca>; david.macarthur@ec.gc.ca; Birch, Angela <Angela.Birch@novascotia.ca>; Cottreau-Robins, Catherine M <Catherine.Cottreau-Robins@novascotia.ca>; Blackburn, Lori M <Lori.Blackburn@novascotia.ca>; Boudreau, Louise O <Louise.Boudreau@novascotia.ca>; Gautreau, Rachel (EC)

Cc: Clancey, Lewis <Lewis.Clancey@novascotia.ca>; Ceschiutti, Robert <Robert.Ceschiutti@novascotia.ca>
Subject: FW: AQ#1442, 1443, 1444 - New Application

Attn: Network Review Agencies:

Please be reminded that our office has not received comments from your Department for the proposed aquaculture site in Antigonish Harbour, Antigonish County. Your comments are due on or before **December 7, 2020.**

Megan Greenwood

Licensing Coordinator
Nova Scotia Department of Fisheries and Aquaculture
1575 Lake Road
Shelburne, NS B0T
1W0
Phone 902-875-7443
Fax 902-875-7429

From: Greenwood, Megan N <Megan.Greenwood@novascotia.ca>
Sent: December 4, 2020 2:20 PM
To: Ceschiutti, Robert <Robert.Ceschiutti@novascotia.ca>**Subject:** Fwd: Request for extension AQ#1442, 1443, 1444 - New Application

Sent from my iPhone Begin

forwarded message:

From: "Boudreau, Louise O" <Louise.Boudreau@novascotia.ca>
Date: December 4, 2020 at 2:17:14 PM AST
To: "Greenwood, Megan N" <Megan.Greenwood@novascotia.ca>
Cc: "O'Brien-Latham, Lesley" <Lesley.OBrien-Latham@novascotia.ca>, "Blackburn, Lori M" <Lori.Blackburn@novascotia.ca>
Subject: Request for extension AQ#1442, 1443, 1444 - New Application

Hello Megan,

Thanks so much for the reminder. I'm working on consolidating the comments I have received on these three applications. It would be helpful to have a bit more time for my director and our biology reviewer to approve the comments.

Would it be possible to get a bit more time? I don't want to rush anyone.
Wednesday EOD would be helpful but I may be able to get it to you earlier.

Warm Regards,

Louise

Louise Boudreau

Policy Analyst

Department of Lands and Forestry
Founders Square |1701 Hollis Street, 3rd Floor | Halifax, NS B3J 2T9 |
424-3530

From: Ceschiutti, Robert <Robert.Ceschiutti@novascotia.ca>
Sent: December 4, 2020 4:11 PM
To: Boudreau, Louise O <Louise.Boudreau@novascotia.ca>
Cc: Greenwood, Megan N <Megan.Greenwood@novascotia.ca>; Clancey, Lewis <Lewis.Clancey@novascotia.ca>
Subject: FW: Request for extension AQ#1442, 1443, 1444 - New Application

Hi Louise, thanks for the heads-up, I approve of an extension for response to the following week, Monday December 14th, 2020.

Regards,
Robert Ceschiutti
Manager, Licensing and Leasing
NS Department of Fisheries and Aquaculture
1575 Lake Road
Shelburne, Nova Scotia
B0T 1W0
Phone: 902-875-7430
Cell: 902-874-0996
Robert.Ceschiutti@novascotia.ca

From: Boudreau, Louise O
To: Ceschiutti, Robert
Cc: Greenwood, Megan N; Clancey, Lewis
Subject: RE: Request for extension AQ#1442, 1443, 1444 - New Application
Date: December 4, 2020 4:13:24 PM

Hi Robert,

Thanks so very much. The comments that I'm reviewing need a bit of work and I want to go back to the reviewer. I also want to give my director enough time.

You are a life saver!

Louise

Louise Boudreau
Policy Analyst

Department of Lands and Forestry

Founders Square |1701 Hollis Street, 3rd Floor | Halifax, NS B3J 2T9 |
424-3530

Network Agency Review of an Aquaculture Application

Agency	**Lands and Forestry**
Division (if applicable)	**Policy Division on behalf of the Department Lands and Forestry**
Date	**Dec. 11th 2020**
File No.	**1444**
Type of application	**New Marine Shellfish**
Information Provided	

Please provide comments, concerns, recommendations, or requirements on the above stated application for a marine aquaculture licence. Please include the criterion /criteria within your jurisdiction or mandate that your feedback is based upon. Similarly, if additional information is required to make a determination, please include the criterion /criteria within your jurisdiction or mandate that your request is based upon.

☐ No concerns regarding the proposed development
☐ Concerns with development are expressed below
☒ Request modifications to the proposed development (described below)
☒ Required or recommended conditions (described below)
☒ Request additional information (described below)
☐ Request meeting with applicant and NSDFA (described below)
☐ No comments on the application

Comments, concerns, recommendations, and/or required conditions including the criterion /criteria within your jurisdiction or mandate that your feedback is based upon. (Attach comments if preferred, or add additional pages, as required.):

Public Notice and Disclosure

As part of the process for deciding on an application, it may be necessary for the Nova Scotia Department of Fisheries and Aquaculture (“Fisheries and Aquaculture”) to disclose the collected network review information to the applicant and other government bodies, including, if applicable, the Nova Scotia Aquaculture Review Board for use at an adjudicative hearing relating to the application in question.

In accordance with departmental policy, which seeks to promote public involvement in the process for deciding on aquaculture applications, Fisheries and Aquaculture will disclose aquaculture application information, including network review information, on the departmental website.

Privacy Statement

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COMMENTS

The Department of Lands and Forestry has the following comments:

Crown Land:

According to the records on file at the Crown Land Information Management Centre this project would not require approvals/permits/authorities from Land Administration since any land lying below the original ordinary high water mark of Antigonish Harbour, Antigonish County, at the location provided, is considered ungranted Crown land with no encumbrances. It should be noted that the scope of this research only incudes information on file at this office relating to Nova Scotia Lands and Forestry ownership and anything affecting that interest.

Parks and Beaches Act:

The Department has the following concerns:

1. The proposed lease is within the protected beach boundary of Dunns and Monks Head Protected Beach and adjacent to a pending provincial park: Dunns Provincial Park Reserve. Recreation is a mandate of the Department under the Parks and Beaches Act. The area is known for recreational activities such as boating, fishing, and kayaking. The beach is appealing for recreation as it is sheltered and generally somewhat calm. If this application is approved, this project would potentially remove the ability for the public to use this portion of the beach. Meaningful consultation with recreational users to better understand recreational values needs to take place to address concerns and mitigate impacts. .
2. The proposed 230-meter buffer from the dune crest to the edge of the farm appears to be acceptable (pending results from fisher, recreational users, Mi'kmaq consultation, and biological studies). However, if this project is approved, the Department will need to review the area every 5 years to account for coastal migration, changes to the beach system, ongoing monitoring, and unanticipated impacts. This may require that the lease location be moved.

Wildlife/Wildlife Habitat:

Request for Additional Information:

The Department does not have sufficient information to complete a fulsome review of this application. The Department has concerns regarding the lack of evidence in the document including but not limited to baseline data, scientific references or evidence, and lack of mitigation for wildlife issues. There is no evidence that this project will not contribute to negative impacts to the local ecosystem and wildlife. There are only a few instances of the proponent outlining a possible negative impact of the project and providing mitigation. More information is needed to determine if there will be more negative impacts resulting from this project and if more mitigation is needed.

Aquaculture lease 1444 is adjacent to multiple biodiversity features:

1. Dunn's beach Provincial park is a provincially protected beach.
2. Designated SIGHAB and Federal Critical Habitat for the endangered Piping Plover (PIPL)- Dunn's beach is a known piping plover breeding area. Significant effort has been put forward to research and provide an adequate setback from Piping plover SIGHAB and Critical Habitat as per the application. There is limited scientific literature on the effects of noise/oyster cultivation on breeding success of PIPL. It is well known that human foot traffic, dogs, vehicle use, and habitat loss are the main factors effecting PIPL. This project does not contain any of those stressors provided the proponent is restricted from any on-land activity in the vicinity of PIPL habitat.
3. The lease is situated in behind Dunn's beach. This is a staging area for many types of sea birds (cormorants, ducks etc.). There is potential for this project to reduce the ability for birds to stage etc. due to boat traffic/human disturbance. However, there are other staging areas in the immediate vicinity. This may also have unwanted impacts on the quality of oysters. More information needed.
4. There are 2 wetlands of special significance to the east ~200m away from the lease. No negative effects from this project are anticipated.

The Department of Lands and Forestry has the following comments:

Comments on the Application Document

1. The Department is concerned that the application document indicates that there will be some changes to the local environment (negative or positive) but does not clearly identify those changes. It is also a concern that potential negative impacts are not considered or dismissed and that no mitigation is put forward for these potential impacts.
2. There is an overall lack of baseline data which is necessary to monitor the project going forward. Since this project does not provide baseline data, mitigation, or data to support the safety of this project, it can not be determined if this oyster farm could cause harm to Species at Risk and other species protected through the *NS Wildlife Act* and the *Migratory Birds Convention Act*.
3. The document relies on repetitious data, individual opinions, and anecdotes. It relies on assertions made by individuals, many whose names are redacted and without supporting evidence. The document provides almost no evidence for its assertions on wildlife. The Piping Plover report is an exception and provides adequate detail on the species and how mitigation was incorporated. The overall quality of the document does not meet the information needs of the Department, and lacks structure making it difficult to navigate

(no table of contents, page numbers, references, many repetitions, and no background data/preamble). There are many instances indicating that literature reviews were completed but the report fails to provide evidence of this work.

4. The report is largely void of biological data, evidence, or baseline data (other than Piping Plover) necessary to assess the environmental impact.

Recommendations

The Department does not have sufficient information to fully assess this proposal and provide recommendations for mitigation measures concerning the potential biodiversity impacts. Specifically, the Department requires:

1. Base line data and analysis that identifies the potential negative impacts on the ecosystem due to the increased load of shellfish. Ecosystem level impacts include negative impacts that could potentially harm the aquatic ecosystem and species (vegetation, benthic species etc.) and have consequent impacts on the food chain
2. A comprehensive wildlife management plan that identifies impacts on birds is required. This was identified as a potential issue in the application. The plan must identify which species could be impacted, how these impacts can be mitigated and how to monitor for incidental impacts (entanglement, disturbance during resting periods, noise issues etc.). A thorough risk assessment is required to create this wildlife management plan.
3. Baseline data for benthic invertebrates, vegetation, water quality etc. This is necessary to ensure the long-term sustainability of the project and harbour ecosystem. This data should be collected, and a monitoring plan should be created to adequately monitor environmental impacts; positive or negative.
4. Information (literature review, pre-disturbance data etc.) collected or collated on the following issues:
   a. Effects of de-fouling on the local water quality, species etc.
   b. Noise from farm- effects on birds and any other potentially impacted wildlife (A literature review may suffice).
   c. Impacts for birds etc. where potential impacts may be caused by suspended oysters, noise, and the effects bird waste has on oyster quality if nearby.
   d. Possible effects on eelgrass and baseline data on current extent of eelgrass.

Recommended Conditions of Approval

The proponent must develop a wildlife management plan that is acceptable to the Department of Lands and Forestry. The proponent is required to implement the wildlife management plan as approved.

The proponent must:

1. Not disturb, harass, or chase congregated birds (waterfowl, geese, cormorants etc.), especially during January – March.

2. No perform any work on land on Dunn’s beach and any of the corresponding park land.
3. Remove any gear/equipment that is washed ashore/deposited on any Crown land at their expense.
4. Report all wildlife mortalities (entanglements etc.) to the Department of Lands and Forestry Regional Biologist and any other appropriate agency (if a marine mammal, Species at Risk etc.) within 5 business days.

Town Point Consulting Inc.

March 3, 2021

NS Department of Fisheries and Aquaculture
1575 Lake Road
Shelburne, NS
B0T 1W0

Attention; Nathaniel Fiendel, Lewis Clancey

Re: AQ1442,1443,1444

Please see below our reply to the Network Agency Review received from Dept of Lands and Forestry.

Lands and Forestry

1) Draft Wildlife Management Plan

Antigonish Harbour is a vibrant and valuable ecosystem that provides important habitat for many species of wildlife. It is a known stopover point for migratory birds, numerous species of shore birds forage and nest on the adjacent beeches, salmon and salmonoids transit through the harbour on their way to and from the tributary rivers to mention just a few functions this estuary provides. This estuary is typical of the "drowned riverbed estuaries found along the south west shore of the Gulf of St Laurence. They are shallow water systems (mostly 1-5m) with sand dune barrier beaches and with soft-bottom communities typically dominated by Zostera marina L." (Watt, Garbary & Longtin 2009)
Given the ecological value of these estuaries it is important to ensure that activities within them can be conducted without serious disruption of those species dependant on the estuaries for their reproduction, feeding, and other life cycle functions. While there has been no prior study of Antigonish Harbour related to the interaction of oyster aquaculture operations and wildlife the following study is relevant and applies to a similar estuary in the gulf region.
Habitat Management Qualitative Risk Assessment: water Column Oyster Aquaculture in New Brunswick 2007 Daigle, Hardy & Robichaud
"An Ecological Risk Assessment and a Net Ecological Benefit Analysis are used to make determinations as to the effects and functions, respectively, of water column oyster aquaculture in gulf NB. Using the risk assessment, we conclude that the overall "scale of potential negative effects" of water column oyster aquaculture and the "sensitivity of fish and fish habitat" correspond to low-risk activity which is not likely to significantly harm the productive capacity or the ecological integrity of fish habitat. Moreover, our analysis suggests that oysters in aquaculture can potentially be of significant benefit to these estuaries and can help to restore many important ecological functions which were reduced following the historical decline of natural populations."
"The geographic area for which the risk assessment was needed in Gulf New Brunswick (N.B.), but could also apply to Prince Edward Island (P.E.I.) and gulf Nova Scotia (N.S.)."
This study is lengthy but essentially it concludes that off-bottom oyster aquaculture operations do not impose significant negative impacts on fish and fish habitat. Reasons for this include the following.
"Some authors have proposed that the aquaculture equipment itself, and other structures, may

contribute to estuarine productivity by creating hard substrate…" "Aquaculture gear increased habitat complexity and supported higher abundances of organisms than non-vegetated seabed: this was determined to be particularly beneficial to recreational and commercial fish and invertebrate species in their early life stages. DeAlteris et al. (2004) concluded that the relative habitat value of aquaculture gear is at least equivalent to submerged aquatic vegetation."

The referenced study should satisfy concerns related to fish, aquatic plants, and invertebrates. Potential impacts of suspended oyster aquaculture on seabirds and shore birds has been less studied maybe because these species have not demonstrated significant impact from suspended oyster aquaculture. However, the study titled **Bivalve aquaculture in estuaries: Review and synthesis of oyster culture effects** by Forrest et al relates in part interaction of raised intertidal oyster aquaculture gear with seabirds and shore birds. Granted intertidal gear is not the same as suspended aquaculture gear but some of the interactions may be similar.

The study states "In contrast, the few other published studies directly investigating interactions between elevated oyster culture and birds provide little evidence for significant adverse effects." Also stated is "When the range of effects is considered as a whole it could be argued that some nominally adverse effects may be compensated to some extent by more positive effects. For example, although natural seabed sediments and benthos may be altered beneath cultivation structures, benthic production may increase. Together with the creation of novel habitat, such changes may benefit some fish and bird species and provide a range of other beneficial ecosystem services such as local enhancement of biodiversity."

Another study called **Oyster farming and shorebirds likely can coexist** from Rutgers University states "the study showed foraging rates were mostly influenced by environmental conditions, especially the presence of gulls or other shorebirds. None of the four bird species of concern substantially altered their foraging behavior due to the presence of tended or untended oyster aquaculture." Given that this study only considered intertidal operations and the proposed farm is suspended and not intertidal the proposed farm would have much greater separation from foraging shorebirds so logically the impact would be even less or totally non-existent.

The greatest opportunity for interaction between birds and the proposed farm is a result of roosting on the floating growth units and associated bouys. We see no issue with birds roosting on bouys associated with this operation. The opportunity to roost on these structures may even provide benefit to the birds. The issue of birds roosting on growth units is however a concern from a farm operation perspective because as they roost, they defecate onto the oysters below. This is a potential food safety issue and it presents a public perception challenge for the farmer and the industry. Our development of the BOBR growth unit, which will be used on this farm, considered this problem. BOBR growth units are almost neutrally buoyant and cylindrical in shape. This combination results in a very unstable perch for birds and insufficient buoyancy to support heavy birds such as cormorants. In seventeen months of trials at ShanDaph Oyster farm in Merigomish Harbour with BOBR deployed adjacent to Oyster Gro and other cage types no cormorants were observed roosting on BOBR units while they commonly roosted on the adjacent gear. Furthermore, the negative image of accumulated bird feces cannot happen on BOBR because the entire unit is always within the splash zone so if a bird were to roost on BOBR it would not result in the unsightly mess that leads to this aspect of poor perception.

We believe the use of BOBR growth units largely mitigates the problem of interaction between seabirds and floating growth units. As for the possibility of entanglements or predation of stock by birds we consulted Robin Stewart, aquaculture consultant to First nations groups locally and in Cape Breton. He confirmed no known instances of bird entanglement associated with suspended oyster aquaculture operations during his 40 years of industry involvement. When asked why he said, "birds don't eat oysters" and even if they did, they cannot access the oysters through the Vexar bags used in BOBR and Oyster Gro cages. The problem with birds seeking a food source at shellfish aquaculture sites relates to mussels not oysters.

Dr. Tony Miller professor emeritus from St FX came to our site on Wednesday Feb. 24, 2021. Tony is a biology prof who studied Antigonish Harbour extensively for many decades and frequently canoes there to study birds.
Tony came to my property on Wednesday for the afternoon and we spent 31/2 hours discussing our farm plan and its potential impact on birds.

Points of note from this discussion are as follows;

1) Regarding Plovers on Dunn's Beach, Dr. Miller agrees they use the ocean side not the harbour side of the beach and he did not see any problem with the proposed operation relative to plovers. He identified the area used by Turns which is near the western tip of the beach, also sufficiently away from the lease sites.

2) Regarding the Captain's Island site, Dr. Miller had no concerns. He brought up geese repeatedly but always pointed out that they use the area between Captain's Island and Mahoney's Beach. Dr. Miller said several times that he sees no reason our plan would present a problem for birds that use the estuary.

Regarding disturbance during resting periods, farm operations are daytime only. Only during an emergency or some other unusual circumstance would farm boats be present on the lease sites after dark. As for noise issues, we have revised the design of the farm service vessel to remove the hydraulic system associated with the tumble function and now rely only on the much quieter outboard motor to provide the mechanical effort for this function. There will be a small HP motor to service the water pump necessary for washdown functions. It will be enclosed, well muffled, four stroke and operated only when water supply is necessary. The outboard motor will be well muffled, four stroke, and all motors will be maintained to ensure mufflers are properly functioning. Crews will be instructed not to rapidly approach congregations of seabirds and not to chase or harass birds.
Other wildlife management plan items intended to mitigate possible negative impacts on wildlife in or near the proposed lease areas are as follows.

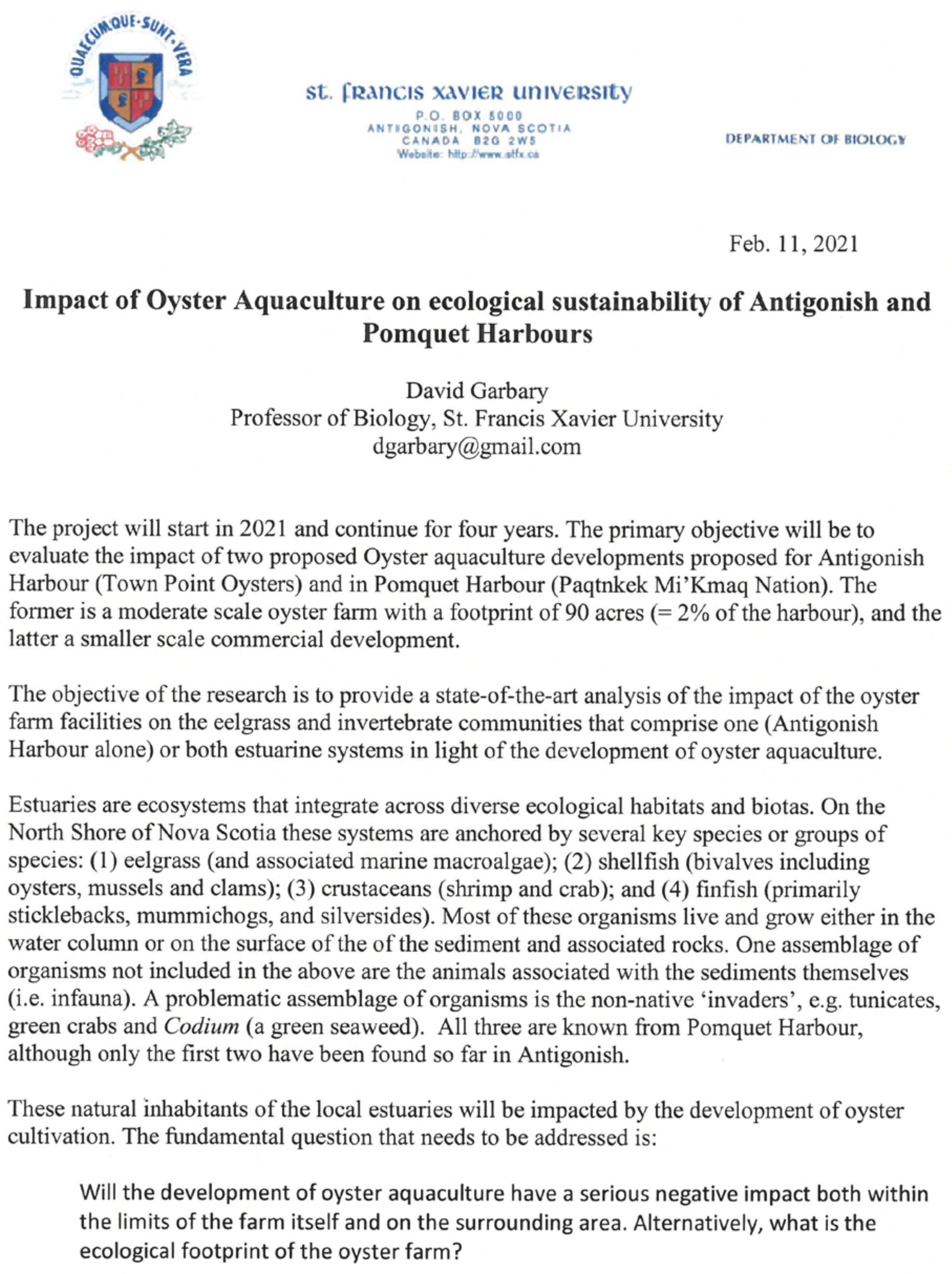
- Ensure no litter, including food scraps, is left or deposited in the coastal area.
- Ensure staff are aware of the importance of avoiding female eiders with brooding chicks, groups of moulting eiders or flocks of staging or wintering birds. Vessels will reduce speed when in the vicinity of flocks of birds.
- Ensure staff are well versed in the requirements of the Migratory Birds Convention Act and associated regulations.
- Staff will be instructed not to disturb, harass, or chase congregated birds (waterfowl, geese, cormorants, etc.) especially during January – March.
- No work will be performed on land on Dunn's Beach and any of the corresponding parkland.

- Report all wildlife mortalities associated with the proposed operation to the Department of Lands and Forestry regional biologist, Harrison Moore, and any other appropriate agency within 5 business days.
- Place lines and configure units so that long lines run parallel to tidal currents and prevailing winds to avoid overlap with the same area of vegetation especially eelgrass. See farm layout drawings included in the application.
- Space rows of growth units a minimum of 3m apart and not to exceed 50% coverage of the surface area of the lease.
- Locate off-bottom aquaculture structure away from high density (meadow) areas of eelgrass cover. The study called **Methods for Mapping and Monitoring Eelgrass habitiat in British Columbia by Environment Canada** states "a dense meadow of intertidal eelgrass may have a density of 2000 shoots/m^2" Dr. Barrell with DFO told us during a site visit the areas we have proposed are not a meadow because the shoot density is in the 10's/m^2. Please contact him to discuss or access his report …we don't have it to share.
- Respect environmental and social standards as demonstrated by the Best Aquaculture Practices Certification Standards, Guidelines for growing sites in Nova Scotia.
- Minimize sedimentation of the waterbody during all phases of the installation, operation, and maintenance of the aquaculture facility.
- Design and install structures to maximize openings to increase light penetration.
- Develop and implement a response plan to avoid a spill of deleterious substance.
- Identify aquatic invasive species of concern and report any sightings to DFO.
- Carry out a post monitoring survey (1,3 and 5 years) including photos and video and characterization of eelgrass vegetation within the aquaculture lease boundary and at a reference site outside of the lease boundary. This will be done as part of the StFX study.

2) Baseline data for benthic invertebrates, vegetation, water quality etc.

We have been working with Dr. Garbary, head of the Aquatic Resources program at St FX to address this issue in a broad and comprehensive manner. Below is the proposed study program we plan to implement which should more than satisfy any related concerns. Essentially the study involves a pre-farm survey to establish pre-farm conditions and then a three-year follow-up program to confirm actual impacts, positive and negative. This is valuable science that will contribute to the knowledge base related to environmental impacts of oyster aquaculture. Our involvement in the study will enable real-time awareness of impacts allowing timely mitigation measures should the need arise.

The initial outline of this collaborative study between StFX, Town Point Oysters and potentially P is as follows;



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ANTIGONISH, NOVA SCOTIA
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DEPARTMENT OF BIOLOGY

Feb. 11, 2021

Impact of Oyster Aquaculture on ecological sustainability of Antigonish and Pomquet Harbours

David Garbary
Professor of Biology, St. Francis Xavier University
dgarbary@gmail.com

The project will start in 2021 and continue for four years. The primary objective will be to evaluate the impact of two proposed Oyster aquaculture developments proposed for Antigonish Harbour (Town Point Oysters) and in Pomquet Harbour (Paqtnkek Mi'Kmaq Nation). The former is a moderate scale oyster farm with a footprint of 90 acres (= 2% of the harbour), and the latter a smaller scale commercial development.

The objective of the research is to provide a state-of-the-art analysis of the impact of the oyster farm facilities on the eelgrass and invertebrate communities that comprise one (Antigonish Harbour alone) or both estuarine systems in light of the development of oyster aquaculture.

Estuaries are ecosystems that integrate across diverse ecological habitats and biotas. On the North Shore of Nova Scotia these systems are anchored by several key species or groups of species: (1) eelgrass (and associated marine macroalgae); (2) shellfish (bivalves including oysters, mussels and clams); (3) crustaceans (shrimp and crab); and (4) finfish (primarily sticklebacks, mummichogs, and silversides). Most of these organisms live and grow either in the water column or on the surface of the of the sediment and associated rocks. One assemblage of organisms not included in the above are the animals associated with the sediments themselves (i.e. infauna). A problematic assemblage of organisms is the non-native 'invaders', e.g. tunicates, green crabs and *Codium* (a green seaweed). All three are known from Pomquet Harbour, although only the first two have been found so far in Antigonish.

These natural inhabitants of the local estuaries will be impacted by the development of oyster cultivation. The fundamental question that needs to be addressed is:

> Will the development of oyster aquaculture have a serious negative impact both within the limits of the farm itself and on the surrounding area. Alternatively, what is the ecological footprint of the oyster farm?

The extent to which negative effects will occur needs to be understood. Only by defining the impacts can one then make appropriate farm management decisions, e.g, to mitigate impacts, or to accept these impacts as minor perturbations of the local ecology.

It is also possible that oyster aquaculture will have positive impacts on the estuarine ecosystems. Some of these potential benefits include the following:

(1) creating habitat diversity in and around the oyster cages;
(2) water clarification than can increase the overall production of eelgrass; and
(3) removal of a major eelgrass pathogen EWD (a slime mold) that destroyed eelgrass populations in the 1930s.

The fact that Paqtnkek Mi'Kmaq Nation will be using a combination of traditional oyster grow cages and the new technology (BOBR growth units) devised by DockPort Ltd, a sister company to Town Point Oysters, provides a unique research opportunity. Thus, this research program will compare the impacts of the two technologies in one estuary (i.e. Pomquet Harbour) and then provide a realistic comparison of oyster cultivation in two adjacent estuaries of St. Georges Bay: Antigonish and Pomquet Harbour. The proposed Town Point farm will use only BOBR growth units.

This proposal is for a four-year study to evaluate an ecosystem prior to the start of oyster cultivation, to evaluate it during the ramping up of oyster cultivation, and then to evaluate the ecosystem at full production. Over this time period we will:

1) Evaluate the state of the eelgrass beds and associated macroalgae within the farm areas and in adjacent areas (up to six sites in each harbour). This will require studies of growth and reproduction of eelgrass at the various sites. We will monitor both above-ground and below-ground biomass of the eelgrass. This evaluation will determine the extent of negative impact to eelgrass within the farm sites, the extent of positive impact beyond the farm sites and an assessment of the net benefit to the estuary relative to eelgrass.
2) Monitor eelgrass wasting disease to determine if oyster cultivation has a positive impact by reducing the impact of the systemic parasite that causes the disease.
3) Examine the state of invertebrate fauna (i.e. shrimp, crabs, snails, etc.) at the study sites along with animals that live in the sediments.
4) Quantify the distribution and abundance of invasive tunicates, green crabs and *Codium* in the both harbours.
5) Compare the impacts of two different oyster grow technologies; and
6) Conduct water sampling for turbidity, chlorophyll, to determine if farm operations provide a positive impact on water quality.

Draft budget:

1) Funding of undergraduate students for the period May to August @ $6000 per student. Total $12,000 per year for four years = $48,000.

2) Funding for graduate student support @ $20,000 per year per student beginning in year two. Ideally two graduate students will be involved, one to focus on the eelgrass and seaweed side of the research; the second to focus on the animal side of the research. $40,000 per for three years = $120,000

3) Travel will include trips to Antigonish Harbour and Pomquet Harbour at least twice per week during summer @$1500 per year for four years = $6,000

4) Collecting equipment and preservation of samples, $1000 per year = $4,000

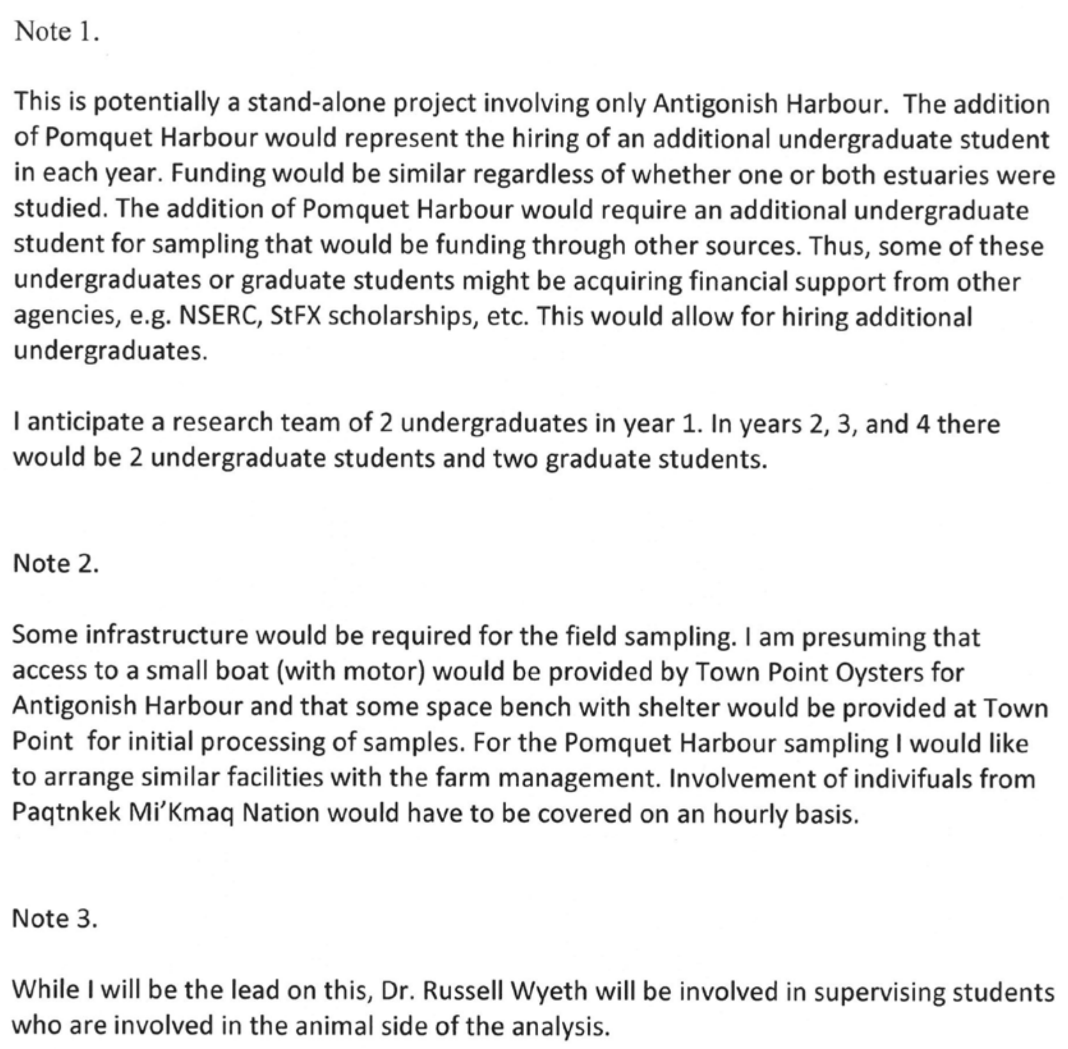
5) YSI Chlorophyll/turbidity/temperature meter = $13,692

6) Subtotal = $191,692

7) Farm operator general cost for fuel, boat use, support labour 10% of above = $19,169

8) University overhead @ 25% of the above = $52,715

Total = $263,576 @$68,594 per year.



Note 1.

This is potentially a stand-alone project involving only Antigonish Harbour. The addition of Pomquet Harbour would represent the hiring of an additional undergraduate student in each year. Funding would be similar regardless of whether one or both estuaries were studied. The addition of Pomquet Harbour would require an additional undergraduate student for sampling that would be funding through other sources. Thus, some of these undergraduates or graduate students might be acquiring financial support from other agencies, e.g. NSERC, StFX scholarships, etc. This would allow for hiring additional undergraduates.

I anticipate a research team of 2 undergraduates in year 1. In years 2, 3, and 4 there would be 2 undergraduate students and two graduate students.

Note 2.

Some infrastructure would be required for the field sampling. I am presuming that access to a small boat (with motor) would be provided by Town Point Oysters for Antigonish Harbour and that some space bench with shelter would be provided at Town Point for initial processing of samples. For the Pomquet Harbour sampling I would like to arrange similar facilities with the farm management. Involvement of indivifuals from Paqtnkek Mi'Kmaq Nation would have to be covered on an hourly basis.

Note 3.

While I will be the lead on this, Dr. Russell Wyeth will be involved in supervising students who are involved in the animal side of the analysis.

3) Information

a) Effects of de-fouling on local water quality, species etc.

TPO does not intend to use chemical de-fouling, only natural processes such as desiccation or heated seawater dipping for de-fouling growth units and stock. Therefore, only materials from the harbour will be returned to the harbour through de-fouling operations.

It is our plan to enable more frequent de-fouling of BOBR units than is practical with Oyster Gro or similar systems by using our service vessel called "Oyster-Matic" which is part of the BOBR growth system developed by our sister company DockPort Ltd. This system mechanizes the common husbandry tasks of tumbling, sorting, de-fouling, harvesting, sinking, and raising the BOBR growth units. Because the time and cost associated with each of these tasks will be comparatively much lower than the same tasks with competing systems frequency of operations may be increased without adversely effecting the input production costs. More frequent de-fouling and tumbling will produce a superior product that should get to market more quickly.

The de-fouling process will involve pressure washing and a 12 second heat dip. This operation will be conducted on the lease sites without the need for removing bags from the lines.

The removed fouling organisms with be returned to the water as is the case with Oyster Gro and most other systems. Regarding the impact of this returned material on the benthic environment, species etc. the following article addresses the matter.

The effect of floating bag management strategies on biofouling, oyster growth and biodeposition levels by A. Mallet et al

This study states “Overall, there was no indication that floating bag oyster culture, even in cases where the bags were heavily fouled, significantly increased biodeposition levels relative to the reference sites.”

The study titled **Habitat Management Qualitative Risk Assessment: Water Column Oyster Aquaculture in New Brunswick** By Daigle, Robichaud & Hardy states the following on page 40.

“In the case of water column aquaculture, studies on sedimentation rates in St. Simon Bay N.B. showed that deposition rates increased at culture sites possibly from the oysters, fouling organisms and hydrodynamic effects of equipment (Mallet et al 2006). However, the mean organic content of the sediment deposited at the oyster table site (20.2%) was not significantly different from the Floating Bag (21.8%) or the Reference sites (21.8%) (Mallet et al 2006). The authors suggest that the lack of enrichment of the sediments indicated that the organic matter in the bio deposits was not being incorporated into the sediments and was either washed away and/or rapidly processed by the benthos community.”

On page 41 “Therefore, there is no indication to date of significant or adverse effects associated with the increase in biodeposition under water column oyster aquaculture sites in N.B.”

Regarding the effect of de-fouling operations on local water quality, no studies specific to this topic were found, perhaps because it has not been identified as a serious concern. This particular task is occasional not continuous, however the ongoing beneficial effect from oysters throughout the farm filtering and clarifying the water may be considered to be more substantial than the occasional and isolated effect of the de-fouling process. The following article refers to these effects beneficial to water quality.

Modelling carrying capacity of bivalve aquaculture: a review of definitions and methods, Filgueira et al

“For example, it is predicted that the increase of water clarity from bivalve aquaculture and/or oyster restoration may lead to an increased biomass of submerged aquatic vegetation (Newell and Koch 2004; Cerco and Noel 2007; Wall et al. 2008). A recent study conducted in Atlantic Canada showed a positive relationship between farmed oyster biomass and eelgrass (Zostera marina) biomass (Andrea Locke, personal communication). In addition to the direct effect on benthic habitat caused by the proliferation of submerged aquatic vegetation, a reduction of phytoplankton production in the water column is expected, due to the direct competition for nutrients (Souchu et al. 2001; Newell 2004; Porter et al. 2004).”

b) Noise from farm – effects on birds and any other potentially impacted wildlife.

The proposed operation is a low intensity operation where the vast majority of operational time involves very slow speed (walking speed or slower) boat travel and typically one boat per lease area. Transiting to and from lease sites to shore involves normal travel speed, however our BOBR growth system allows husbandry tasks to be conducted on the lease sites without moving bags of product to shore. So far fewer shore trips than most farms.

c) Impacts for birds etc. where potential impacts may be caused by suspended oysters, noise, and the effects bird waste has on oyster quality if nearby.

Potential impacts of suspended oyster aquaculture on seabirds and shore birds has been less studied likely because the associated risks are far less prevalent. However, the study titled **Bivalve aquaculture in estuaries: Review and synthesis of oyster culture effects** by Forrest et al relates in part interaction of raised intertidal oyster aquaculture gear with seabirds and shore birds. Granted intertidal gear is not the same as suspended aquaculture gear but some of the interactions may be similar.
The study states “In contrast, the few other published studies directly investigating interactions between elevated oyster culture and birds provide little evidence for significant adverse effects.” Also stated is “When the range of effects is considered as a whole it could be argued that some nominally adverse effects may be compensated to some extent by more positive effects. For example, although natural seabed sediments and benthos may be altered beneath cultivation structures, benthic production may increase. Together with the creation of novel habitat, such changes may benefit some fish and bird species and provide a range of other beneficial ecosystem services such as local enhancement of biodiversity.”
Another study called **Oyster farming and shorebirds likely can coexist** from Rutgers University states “the study showed foraging rates were mostly influenced by environmental conditions, especially the presence of gulls or other shorebirds. None of the four bird species of concern substantially altered their foraging behavior due to the presence of tended or untended oyster aquaculture.” Given that this study only considered intertidal operations and the proposed farm is suspended and not intertidal the proposed farm would have much greater separation from foraging shorebirds so logically the impact would be even less or totally non-existent.
The greatest opportunity for interaction between birds and the proposed farm is a result of roosting on the floating growth units and associated bouys. We see no issue with birds roosting on bouys associated with this operation. The opportunity to roost on these structures may even provide benefit to the birds. The issue of birds roosting on growth units is however a concern from a farm operation perspective because as they roost, they defecate onto the oysters below. This is a potential food safety issue and it presents a public perception challenge for the farmer and the industry. Our development of the BOBR growth unit, which will be used on this farm, considered this problem. BOBR growth units are almost neutrally buoyant and cylindrical in shape. This combination results in a very unstable perch for birds and insufficient buoyancy to support heavy birds such as cormorants. In seventeen months of trials at ShanDaph Oyster farm in Merigomish Harbour with BOBR deployed adjacent to Oyster Gro and other cage types no cormorants were observed roosting on BOBR units while they commonly roosted on the adjacent gear. Furthermore, the negative image of accumulated bird feces cannot happen on BOBR because the entire unit is always within the splash zone so if a bird were to roost on BOBR it would not result in the unsightly mess that leads to this aspect of poor perception.

We believe the use of BOBR growth units largely mitigates the problem of interaction between seabirds and floating growth units. As for the possibility of entanglements or predation of stock by birds we consulted Robin Stewart, aquaculture consultant to First nations groups locally and in Cape Breton. He confirmed no known instances of bird entanglement associated with suspended oyster aquaculture operations during his 40 years of industry involvement. When asked why he said, "birds don't eat oysters" and even if they did, they cannot access the oysters through the Vexar bags used in BOBR and Oyster Gro cages. The problem with birds seeking a food source at shellfish aquaculture sites relates to mussels not oysters.
Regarding disturbance during resting periods, farm operations are daytime only. Only during an emergency or some other unusual circumstance would farm boats be present on the lease sites. As for noise issues, we have revised the design of the farm service vessel to remove the hydraulic system associated with the tumble function and now rely only on the much quitter outboard motor to provide the mechanical effort for this function. There will be a small HP motor to service the water pump necessary for washdown functions. It will be enclosed, well muffled, four stroke, and operated only when water supply is necessary. The outboard motor will be well muffled, four stroke, and all motors will be maintained to ensure mufflers are properly functioning. Crews will be instructed not to rapidly approach congregations of seabirds and not to chase or harass birds.

d) Possible effects on eelgrass and baseline data on current extent of eelgrass.

Any loss of eelgrass in Antigonish Harbour would be considered a negative impact. An on-site eelgrass survey was conducted by Department of Fisheries and Oceans (DFO) scientists. A DFO scientist noted that the oyster farm will likely have a positive impact on the overall health of the eelgrass in the harbour due to reduced turbidity of the water. There is potential for reduced growth directly under the BOBR growth units due to shading; however, feedback from this DFO scientist noted that the net effect more broadly is likely to be beneficial. Additionally, the design of the growth units TPO will use minimizes the shading of sunlight on the eelgrass below growth units. TPO's growth units will cover only 0.1% of the harbour surface area.

Antigonish Harbour is 4,400 acres. TPO's proposed lease sites comprise 90.3 acres which is 2% of the harbour. However, the growth units occupy less than 4% of lease areas. Therefore, the portion of Antigonish Harbour covered by growth units will be about 0.1%.

An article cited below asserts that eelgrass can benefit from co-culture with oysters, as this co-culture was shown to reduce the severity of Eelgrass Wasting Disease (EWD) by filtering out pathogens that cause EWD. According to this article, operations such as the proposed farm may reduce the chance of disease outbreak by filtering more of the pathogens that cause EWD. EWD is caused by a pathogenic slime mold, "Labyrinthula", which is present in Antigonish Harbour. This pathogen was responsible for the decimation of eelgrass in the 1930"s. Subsequently, stocks rebounded until the early 2000"s when the arrival of the Green Crab, led to another decimation.

"Oysters and eelgrass: potential partners in a high pCO2 ocean" Ecology, Maya L. Groner, Colleen A. Burge, Ruth Cox, Natalie D. Rivlin, Mo Turner, Kathryn L. Van Alstyne, Sandy Wyllie-Echeverria, John Bucci, Philip Staudigel, Carolyn S. Friedman

“In conclusion, our study provides preliminary evidence to suggest that eelgrass and possibly oysters could benefit from co-culture under projected pCO2 conditions”

“Effects of Bivalve Aquaculture on the Environment and Their Possible Mitigation: A Review” Fisheries and Aquaculture Journal, Daria Gallardi

"The functions of water clarification and bio deposition that characterize filter-feeding bivalves are valuable providers of ecological services to shallow water ecosystems. Bivalves help buffer estuaries and coastal ocean waters against excessive phytoplankton blooms in response to anthropogenic loading of nitrogen, counteracting the symptoms of eutrophication; they also remove inorganic sediments from suspension, counteracting coastal water turbidity. The biodeposition created by mussels and oysters, through the creation of sediment anoxic microzones where denitrifying bacteria are promoted, induce denitrification, which also help to counteract eutrophication by returning nitrogen into the atmosphere as inert nitrogen gas [3,8,9,39,40]. Moreover, the enhancement of water clarity due to filtration allows deeper light penetration and therefore can increase the growth of seagrasses that are important nursery habitat for many fish, crustaceans and molluscs; bivalves are therefore capable of enhancing estuarine nursery habitats [9,40,41]. These natural functions of bivalves can be employed in aquaculture not only to mitigate the environmental effects of the culture, but also to create added value and services for the surrounding environment.”

“Bivalve aquaculture and eelgrass: A global meta-analysis” Aquaculture, Bridget E. Ferriss, Letitia L. Conway-Cranos, Beth L.Sanderson, Laura Hoberecht

“These analyses suggest the response of eelgrass to bivalve aquaculture varies depending on eelgrass characteristics, grow-out approaches, and harvesting methods, with potential regionally specific relationships. Questions remain, regarding how this dynamic relationship between eelgrass and aquaculture habitat relates to ecological functions and services in the nearshore environment.”

As discussed, we would welcome a meeting with NSDLF to discuss the concerns they have expressed and to explore ways to mitigate these concerns.

Best regards,
Ernie Porter, P.Eng., President

Town Point Consulting Inc.

From: Clancey, Lewis <Lewis.Clancey@novascotia.ca>
Sent: March 9, 2021 11:47 AM
To: Boudreau, Louise O <Louise.Boudreau@novascotia.ca>
Cc: Feindel, Nathaniel J <Nathaniel.Feindel@novascotia.ca>
Subject: Aquaculture Lease Application #'s 1442, 1443, 1444, Town Point Consulting Inc.

Hi Louise,

I'm the Aquaculture Advisor responsible for the American oyster lease applications listed above at NS Dept. of Fisheries and Aquaculture.

We are wondering if you would be available to meet with the applicant regarding the lease applications for oyster culture in Antigonish Harbour and your departments network agency review comments that we submitted in response to application. . We are targeting a meeting time of sometime next week, 15-20 February, 2021.

The applicant has received your network agency review comments and would like to have an opportunity to discuss the identified issues your department provided in your review.

Would you, and/or a representative(s) from your department, be available to meet with NSDFA and the applicant via phone or Microsoft Teams, sometime next week to discuss the issues raised in your departments network review of this application?

Thank you,

Lew

Lewis Clancey
Aquaculture Development Advisor
NS Dept. of Fisheries and Aquaculture
1800 Argyle St.
WTCC 6th Floor, Suite 604,
Halifax, NS
B3J2R5

902 956 3839
lewis.clancey@novascotia.ca

From: Boudreau, Louise O <Louise.Boudreau@novascotia.ca>
Sent: March 9, 2021 1:08 PM
To: Clancey, Lewis <Lewis.Clancey@novascotia.ca>
Cc: Feindel, Nathaniel J <Nathaniel.Feindel@novascotia.ca>; Moore, Harrison M <Harrison.Moore@novascotia.ca>
Subject: RE: Aquaculture Lease Application #'s 1442, 1443, 1444, Town Point Consulting Inc.

Hello Lewis,

Thank you for contacting me.. I shared your request with our biology reviewer Harrison Moore. There were a number of challenges with these applications. We would like to hold a meeting with you and Nathaniel to discuss these applications without the applicant. We would like to discuss the types/quality of information we are looking for and to hear your thoughts as well. Most days next week look good for me. I'm not sure what days would work best for Harrison.

Warm Regards,

Louise

Louise Boudreau
Policy Analyst
Department of Lands and Forestry
Founders Square |1701 Hollis Street, 3rd Floor | Halifax, NS B3J 2T9 |
424-3530

From: Clancey, Lewis <Lewis.Clancey@novascotia.ca>
Sent: March 11, 2021 11:37 AM
To: Boudreau, Louise O <Louise.Boudreau@novascotia.ca>
Cc: Feindel, Nathaniel J <Nathaniel.Feindel@novascotia.ca>; Moore, Harrison M <Harrison.Moore@novascotia.ca>
Subject: RE: Aquaculture Lease Application #'s 1442, 1443, 1444, Town Point Consulting Inc.

Thanks Louise,

We'd be happy to meet with you and Harrison. We'd like to propose Thursday March 18, 2021 between 9:00 and 11:00 AM as time to meet to discuss these files.

Please let us know if that date/time would be convenient for you and Harrison.

Thanks,
Lew

Lewis Clancey
Aquaculture Development Advisor
NS Dept. of Fisheries and Aquaculture
1800 Argyle St.
WTCC 6th Floor, Suite 604,
Halifax, NS
B3J2R5

902 956 3839
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From: Boudreau, Louise O <Louise.Boudreau@novascotia.ca>
Sent: March 11, 2021 11:40 AM
To: Clancey, Lewis <Lewis.Clancey@novascotia.ca>
Cc: Feindel, Nathaniel J <Nathaniel.Feindel@novascotia.ca>; Moore, Harrison M <Harrison.Moore@novascotia.ca>
Subject: RE: Aquaculture Lease Application #'s 1442, 1443, 1444, Town Point Consulting Inc.

Hi Lew,

Thanks so much for accepting our proposal. Next Thursday morning is good for me. Anytime after 9:30. I'll let Harrison respond to let you know if this time works for him as well.

Warm Regards,

Louise

Louise Boudreau
Policy Analyst
Department of Lands and Forestry
Founders Square |1701 Hollis Street, 3rd Floor | Halifax, NS B3J 2T9 |
424-3530

From: Moore, Harrison M <Harrison.Moore@novascotia.ca>
Sent: March 11, 2021 11:41 AM
To: Boudreau, Louise O <Louise.Boudreau@novascotia.ca>; Clancey, Lewis <Lewis.Clancey@novascotia.ca>
Cc: Feindel, Nathaniel J <Nathaniel.Feindel@novascotia.ca>
Subject: RE: Aquaculture Lease Application #'s 1442, 1443, 1444, Town Point Consulting Inc.

9:30 is fine for me.

Harrison Moore, **M.Sc**
Regional Biologist
Lands and Forestry, N.S.
(902)-497-4119

From: Clancey, Lewis
Sent: March 11, 2021 11:42 AM
To: Moore, Harrison M <Harrison.Moore@novascotia.ca>; Boudreau, Louise O <Louise.Boudreau@novascotia.ca>
Cc: Feindel, Nathaniel J <Nathaniel.Feindel@novascotia.ca>
Subject: RE: Aquaculture Lease Application #'s 1442, 1443, 1444, Town Point Consulting Inc.

Great, thanks everyone.

I'll set up a Teams meeting and send out the meeting notice.

Thanks,
Lew

Clancey, Lewis

To: Feindel, Nathaniel J; Boudreau, Louise O; Moore, Harrison M

Subject: **From:** Town Point Oyster Ltd. Network Review Discussion NSL&F

Start: March 18, 2021 9:30:00 AM
End: March 18, 2021 11:30:00 AM

Meeting to discuss review of Town Point Oyster Lease applications.

Microsoft Teams meeting
Join on your computer or mobile app
Click here to join the meeting <https://teams.microsoft.com/l/meetup-join/19%3ameeting_NTFjMTVjMWUtM2JlZS00ZGJmLTgzMTMtNDFmNzFjZWE0NjMz%40thread.v2/0?context=%7b%22Tid%22%3a%228eb23313-ce75-4345-a56a-297a2412b4db%22%2c%22Oid%22%3a%2202f00a57-63df-48e7-8e3d-328bbc173cb2%22%7d>
Join with a video conferencing device
20014895@t.plcm.vc
Video Conference ID: 113 703 507 1
Alternate VTC dialing instructions <https://dialin.plcm.vc/teams/?key=20014895&conf=1137035071>
Learn More <https://aka.ms/JoinTeamsMeeting> | Help <https://support.microsoft.com/en-us/office/join-a-meeting-without-a-teams-account-c6efc38f-4e03-4e79-b28f-e65a4c039508?ui=en-us&rs=en-us&ad=us#ID0EBBAAA=Mobile> | Meeting options <https://teams.microsoft.com/meetingOptions/?organizerId=02f00a57-63df-48e7-8e3d-328bbc173cb2&tenantId=8eb23313-ce75-4345-a56a-297a2412b4db&threadId=19_meeting_NTFjMTVjMWUtM2JlZS00ZGJmLTgzMTMtNDFmNzFjZWE0NjMz@thread.v2&messageId=0&language=en-US>

From: Clancey, Lewis
Sent: April 7, 2021 2:14 PM
To: Moore, Harrison M <Harrison.Moore@novascotia.ca>
Cc: Spencer, Amanda L <Amanda.Spencer@novascotia.ca>
Subject: Town Point Oysters, Antigonish lease application information.

Hi Harrison,

With regard to the Antigonish oyster lease applications and our recent telephone discussion, I've included some further information for your review.

Please see attached.

The proponent, Town Point Oysters, has provided a written response to some of NS Lands and Forestry's concerns and comments, and included a list of citations at the end of the document.

A folder containing zip file copies of the aquaculture papers and studies cited is also attached for your information and review.

Please feel free to call me anytime should you wish to discuss any of the information.

Thanks,
Lew

Lewis Clancey
Aquaculture Development Advisor
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1800 Argyle St.
WTCC 6th Floor, Suite 604,
Halifax, NS
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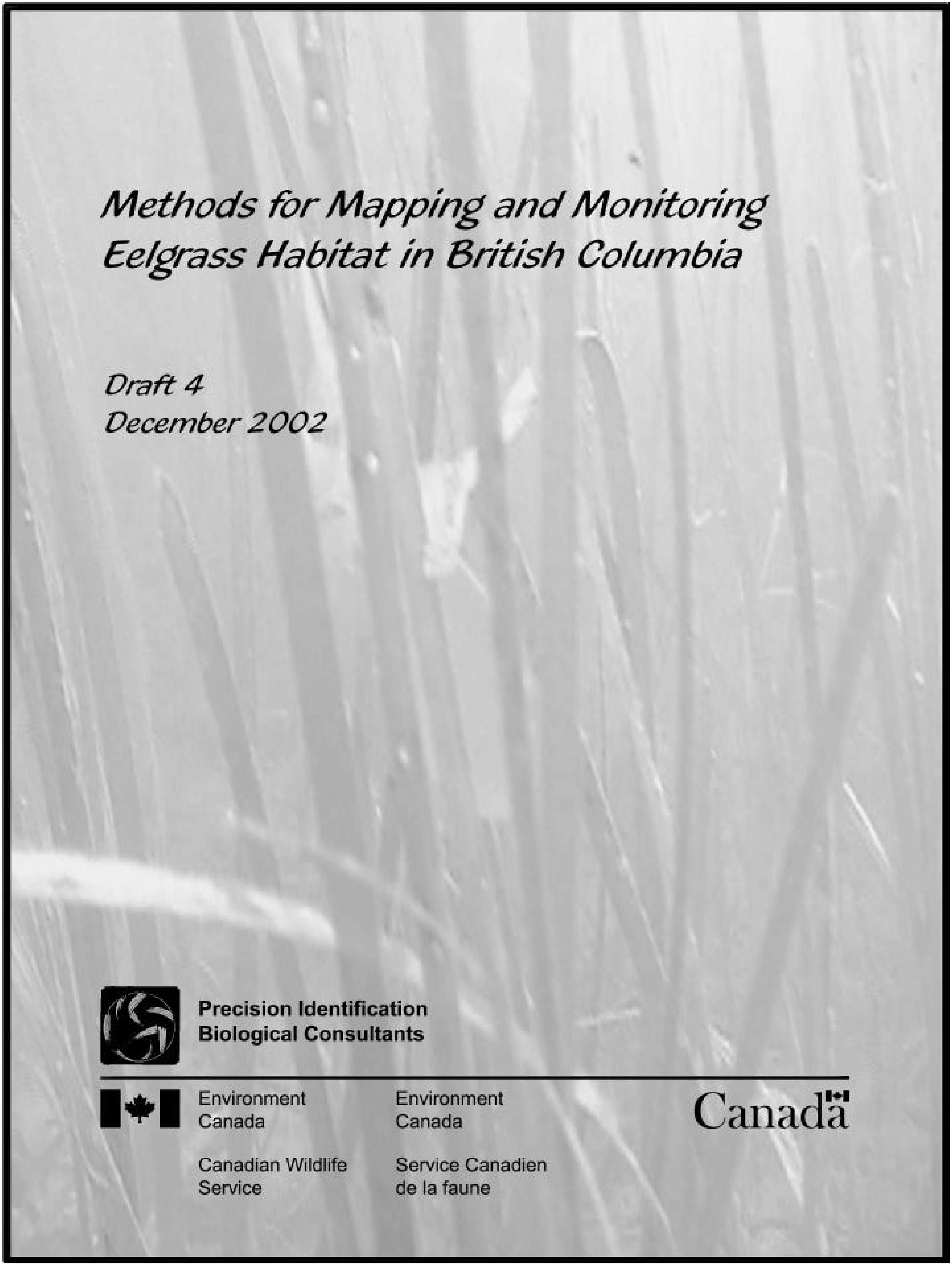

Methods for Mapping and Monitoring Eelgrass Habitat in British Columbia

Draft 4
December 2002

Precision Identification
Biological Consultants

Environment Canada
Canadian Wildlife Service

Environnement Canada
Service Canadien de la faune

Canada

Preface

Field Methods for Mapping and Monitoring Eelgrass Habitat in British Columbia was designed to provide readers with a basic understanding of eelgrass (*Zostera marina* L.) ecology and to provide a standardized set of methods to map, classify, and monitor eelgrass habitat on a local level. The mapping and monitoring system described herein enables community groups and other agencies to contribute consistent and reliable data to a central database.

The manual will be expanded to include a series of monitoring protocols to study various faunal assemblages within eelgrass beds (e.g. fish, zooplankton, and invertebrates). All contributions and comments will be welcomed and acknowledged.

Acknowledgements

The development of this manual was funded by the Canadian Wildlife Service, Environment Canada, Pacific & Yukon Region, Delta; B.C. Jacqueline Booth (Jacqueline Booth and Associates, Salt Spring Island, B.C.) developed the database structure and user interface for storing and retrieving field data on eelgrass beds. She also provided advice based on her experience working with community mapping. Brad Mason (Fisheries & Oceans Canada, Vancouver, B.C.) and Don Chamberlain (Project Watershed, Comox, B.C.) contributed information and guidance relating to the use and limitations of GPS technology. Brad Mason (Fisheries & Oceans Canada, Vancouver, B.C.), Suzanne Richer (Community Mapping Network), and the Community Mapping Network created the internet application and assisted with development of the database structure and functions for providing the online base maps and digitizing tools.

Precision Identification would like to thank and acknowledge the following individuals for their review of the manuscript.

Brad Mason, Fisheries and Oceans Canada, Vancouver, B.C.
Cliff Robinson, Parks Canada, Vancouver, B.C.
Gretchen Harlow, Canadian Wildlife Service, Environment Canada, Vancouver, B.C.
Michele Jones, Mimulus Biological Consultants, Royston, B.C.
Nikki Wright, SeaChange, Victoria, B.C.
Sarah Verstegen, SeaChange, Victoria, B.C.
Rob Butler, Canadian Wildlife Service, Environment Canada, Vancouver, B.C.
Sean Boyd, Canadian Wildlife Service, Environment Canada, Vancouver, B.C.
Ramona C. deGraaf, Dept. of Zoology, University of British Columbia, Vancouver, B.C.
The Seagrass Conservation Group, B.C.

Table of Contents

1.0 Introduction

Land use changes and developments have led to a loss of natural estuarine habitat in British Columbia. Agriculture, forestry, and dredging for commercial and residential development have all contributed to the loss. It is anticipated that the pressure to modify natural estuarine habitat for the development of commercial facilities and residential units within coastal areas will intensify in the near future. It is therefore necessary to identify, classify, quantify, and develop a scientifically defensible management strategy for estuarine habitat in order to protect and maintain these valuable areas.

Eelgrass (*Zostera marina* L.) meadows represent one of the habitat types that are threatened by estuarine development. Various types of disturbance in coastal and estuarine environments have led to a decline in seagrass abundance around the world (Short & Wyllie-Echeverria, 1996). Losses in Chesapeake Bay, United States, have resulted from impaired water quality caused by upland development, agriculture, and shoreline development (Orth & Moore, 1983, Dennison et al. 1993). Pollution induced seagrass declines have been documented in the Mediterranean and along the Atlantic coast of Europe (Nienhuis 1983; Hanekom & Baird 1988; Giesen et al. 1990; Short et al. 1991; DeJong & DeJong 1992; den Hartog 1994).

Seagrasses, including eelgrass, have been used as indicators of nearshore ecosystem health in many areas of the world (Sewell et al., 2002). In Chesapeake Bay, a submerged vegetation monitoring program (eelgrass & freshwater vascular plants) identified a link between decreased productivity within the Bay and degraded water quality from upland watershed activities (Orth & Moore, 1983). The data was used to enact legislation to restrict the activities responsible for the impairment of water quality, which was successful in reversing the trend of vegetation loss (Dennison et al., 1983).

Eelgrass provides critical habitat for numerous species including; outmigrating juvenile salmon (*Oncorhynchus* spp.), Pacific herring (*Clupea harengus*), Dungeness crab (*Cancer magister*), and black brant (*Branta bernicla*) (Norris & Wyllie-Echeverria, 2001). The productivity of eelgrass meadows rivals that of cultivated tropical agriculture (Zieman & Wetzel, 1998). Research in Denmark discovered that detritus, primarily derived from eelgrass, was the basic source of nutrition for animals in Danish coastal waters, and that the historic abundance of fish in Denmark was mainly due to eelgrass (Phillips, 1984). The leaves of eelgrass baffle currents, reducing water velocity and promoting sedimentation. The root-rhizome network forms an interlocking matrix, which binds sediment and restricts erosion (Phillips, 1984).

A study by Helfferich and McRoy in 1978 calculated the U.S. dollar value of eelgrass meadows to be $12,325.00 per acre per year based on its contribution to commercial and recreational fisheries and hunting.

The governments of many countries including the United States, Australia, New Zealand, South Africa, and Britain have recognized the value of seagrass habitat and have implemented seagrass mapping and monitoring programs. These programs involve locating and mapping seagrass communities, usually through analysis of aerial photographs, followed by detailed monitoring of specific sites on the ground. The costs associated with these types of inventories are prohibitive in British Columbia at this time.

Eelgrass has been mapped in several areas of British Columbia, by various groups, using various methods. The majority of the eelgrass mapping information (e.g. herring spawn surveys) was completed in the late 1970s, and may not reflect current conditions.

Environment Canada commissioned the following report to provide the necessary understanding of eelgrass ecology and mapping methodologies to identify, classify, and quantify eelgrass habitat in British Columbia on a local level. The mapping and monitoring system enables local groups and organizations to contribute consistent and reliable data to a central database.

An interactive data entry tool has been developed for this purpose, and is available on the Community Mapping Network website (http://www.shim.bc.ca/eelgrass/main.htm). The data that are collected will be integrated into a larger scale province wide inventory. It is hoped that this information will promote the development of a comprehensive eelgrass mapping and monitoring strategy for British Columbia that may be used to protect eelgrass habitat.

2.0 Eelgrass Ecology

Eelgrass meadows are naturally highly dynamic systems, often changing from year to year or from season to season, reflecting changes in the environment. It is important to understand the natural variability within these ecosystems, in order to avoid false conclusions when assessing changes over time. The following sections were designed to provide an overview of eelgrass ecology and an appreciation for the inherent natural variability both within and between meadows.

Reproduction

Eelgrass reproduces both sexually (seeds) and asexually (branching). The plants flower annually and produce many viable seeds; however very few successfully mature into plants. The flowers are produced on reproductive shoots that develop from vegetative shoots. Once the seeds have developed, the shoot begins to senesce, breaks free from the rhizome, and floats away. Detailed monitoring of eelgrass densities should include enumeration of flowering shoots as well as vegetative shoots, due to the ephemeral nature of the flowering shoots.

Eelgrass reproduces vegetatively by forming new shoots at the base of the parent shoot. The rhizome branches, allowing the new shoot to grow away from the parent shoot. A single plant may have numerous shoots connected via a single branched rhizome. As time passes, older rhizomes decay, so that one plant eventually becomes two or more plants. An eelgrass meadow could, in theory, be composed of many shoots that originated from a single individual.

Species and Ecotypes

There are two species of eelgrass in British Columbia; the native species *Zostera marina* and the introduced species *Zostera japonica*. It is believed that *Z. japonica* was accidentally introduced with oyster spat brought from Japan to aquaculture sites in Washington State (Harrison, 1976). The introduced species is generally smaller and can tolerate exposure (due to its morphology) better than the native species. The introduced species can not compete with the native species due to its smaller size, thus it is not a threat to the native eelgrass. *Z. japonica* is often found adjacent to, or intermixed with, *Z. marina* at higher elevations. The information provided for eelgrass in this document relates specifically to *Z. marina* although it could be easily modified to study populations or meadows of *Z. japonica*.

The leaf length and width of both species varies with depth; as depth increases leaf length and width increases. The leaf length and width of intertidal *Z. marina* is often within the range of *Z. japonica*. Fortunately, the two species have different types of sheaths; this enables one to easily differentiate the species. *Z. marina* has an entire sheath, it is closed to the base; when the lower leaves are slowly pulled in opposite directions the sheath will tear. The sheath of *Z. japonica* is open to the base; thus the sheath parts rather than tears when stress is applied.

It has been proposed that there are races, or ecotypes of *Z. marina* that account for part of the morphological variation (Beckman 1984). It is possible that three of the ecotypes occur in British Columbia. The attributes associated with each ecotype are summarized in Table 1.

Table 1. The habitat and morphological attributes associated with the three ecotypes of *Zostera marina* common in British Columbia. (adapted from Backman, 1984)

Ecotype	Relative leaf size	Leaf width (mm)	Depth range (m)	Seasonal variation in size	Current tolerance
typica	narrow	2 to 5	primarily intertidal	small variation	low
phillipsi	intermediate	4 to15	0 to - 4	large, plant length reduced in winter	moderate
latifolia	large	12 to 20	-0.5 to -10	minimal variation	strongest

An eelgrass meadow may contain one or more ecotype.

The smaller intertidal plants usually occur at a much greater density, due to their smaller size, than those growing in deeper water. For example, a dense meadow of intertidal eelgrass may have a density of 2000 shoots·m^{-2}, while the adjacent subtidal habitat supports 120 shoots·m^{-2}. The biomass (g·m^{-2}) of the less dense subtidal plants can easily exceed that of the intertidal plants due to the larger size of the individual shoots; a factor that must be taken into consideration when sampling.

Cover

The aerial coverage of an eelgrass meadow reflects both the substrate and the hydrodynamic regime. A quiescent environment with a sandy mud substrate generally supports a dense continuous eelgrass bed with virtually 100% cover. The cover of eelgrass in areas subjected to strong currents is typically patchy. Areas with heterogeneous substrate (mixture of fine and coarse) also tend to be patchy.

Eelgrass meadows are spatially dynamic, the edges expand or recede in response to environmental variables. Severe storms may damage or destroy entire meadows. Severe frost (winter) and intense heat (summer) may also kill shoots exposed at low tide. Shifting sand (active sediment bed movement) can have a significant effect on eelgrass distribution.

Density

The density of shoots within an eelgrass bed may be consistent throughout the bed or it may vary in response to environmental parameters within the bed (currents, sediment type, depth, turbidity). In addition, if several ecotypes are present the density will vary depending on the distribution of each ecotype within the bed. In order to determine the mean density of shoots within a bed, the investigator must first establish whether there is any sort of density zonation within the bed, then design a sampling procedure to assess each zone independently. Permanent transects are not recommended as repeated trampling may alter the density along the transect, unless the site is surveyed at high tide using SCUBA or video. Additionally, permanent transect markers collect floating debris and often result in sediment scour.

Environmental Requirements

The growth and distribution of eelgrass is influenced by salinity, sediment type, current velocity, light availability, temperature, and pH. Temperature and pH are not usually restrictive along coastal British Columbia. A summary of the range and optimal levels for each of these parameters is provided in Table 2.

Table 2. Environmental requirements for vegetative growth of eelgrass (Phillips, 1974).

Parameter	**Range**	**Optimum**
salinity	freshwater to 42 ppt	10 to 30 ppt
sediment type	firm sand to soft mud	mixed sand and mud
current velocity	waves to stagnant water	little wave action gentle currents to 3.5 knots
light/depth	1.8 m above MLLW to –30 m	MLLW to – 6.6 m
temperature	-6 °C to 40.5 °C	10 °C to 20 °C
pH	7.3 to 9.0	7.3 to 9.0

MLLW- mean low low water ppt – parts per thousand

The literature reports that eelgrass is restricted to soft sediment; however it is often found in areas with significant amounts of gravel and cobble in British Columbia. There are two known areas where eelgrass has adapted to grow over hard substrate, one on rock in Port McNeil (Durance), and one on cement blocks near Victoria (Austin).

The maximum depth to which eelgrass can grow at a specific location depends on the turbidity of the water, since the amount of light that penetrates the water is reduced when turbidity increases.

3.0 Mapping and Monitoring Parameters

Eelgrass meadows possess many attributes that can be mapped and monitored to assess changes over time and track ecosystem health. The parameters that are selected for study depends on the objectives or goals of the study and the resources available. Monitoring specific meadows, using scientific sampling methods, can provide the data required to detect and assess environmental changes. There are many variables that are commonly measured to detect changes in eelgrass populations or meadows and the environment. The following section reviews the parameters that are frequently used to study eelgrass, and the value associated with each.

Location

An inventory that locates and characterizes eelgrass beds provides a valuable tool that can be used by various resource managers and assist with the development of Integrated Coastal Zone Management plans. Fisheries and Oceans Canada has a policy of 'no net loss', thus proposed development may not impact known eelgrass habitat unless it can be shown that adequate compensation will be provided. Knowing the location of each eelgrass bed would therefore assist in conservation.

Delineation

The delineation of eelgrass beds enables the detection of increases or decreases in area, or range, over time that can be tracked. Losses may be used to detect environmental change, and develop mitigation plans to prevent further degradation. In addition, any industry or development that can be shown to impact eelgrass habitat may be forced by Fisheries and Oceans to provide mitigation, restoration, or compensation.

Depth Distribution

The distribution of eelgrass across a bathymetric gradient is limited at the upper boundary by the degree of exposure at low tide (desiccation) and by light limitations at the lower boundary. In some cases substrate characteristics change with depth; this may also limit eelgrass distribution. Degradation of water quality that results in increased turbidity (e.g. suspended solids, chlorophyll A increases) leads to a decrease in the maximum depth possible for eelgrass survival. Trends in the maximum depth distribution of eelgrass over time can be used as ‘a predictor of ecosystem health’ (Dennison et al., 1983).

Shoot Density

Eelgrass shoot densities vary over time in response to environmental variables (natural and anthropogenic) and are therefore useful indicators of environmental change (Phillips et al., 1983, Olesen et al., 1994). The number of flowering shoots within the meadow is usually determined as part of the density estimate since it may reflect- environmental change or stress, and because the flowering shoots will senesce after they reach maturity, resulting in a decrease of total shoot density.

Distribution

The maximum coverage of eelgrass at a specific site is strongly influenced by the hydrodynamic setting. Quiescent bays tend to support homogenous eelgrass meadows, whereas areas that experience stronger currents and active seabed movement tend to have a patchy eelgrass distribution. The homogeneity of an eelgrass bed can also be reduced by anthropogenic disturbances (shellfish harvesting, boat anchoring, dredging activity, trampling, etc.).

The integrity of an eelgrass bed may be threatened by fragmentation. The plants within established eelgrass beds reduce currents, leading to increased sediment and organic detritus deposition. The dense rhizome and root matrix of the plants, in conjunction with the enhanced deposition rate assists in stabilization of the substrate. ‘If an established, continuous bed becomes fragmented for any reason, the bed will tend to become less stable and more vulnerable to the normal forces of erosion. Channels may form, the cover may become patchier and if the trend continues, isolated patches will develop which are more likely to be washed away. It would appear that there is a threshold of loss, below which destabilization and further losses of beds can occur ‘(Holt et al., 1997).

Monitoring the homogeneity or patchiness of a meadow over time can help to identify impacts and lead to the implementation of mitigation programs to prevent further loss.

Leaf Area Index (LAI)

Leaf area indices are often used to estimate the productivity of eelgrass and the amount of habitat available for colonization by epifauna. The LAI is calculated according to the following formula:

$$\text{LAI} = \text{mean shoot length} \times \text{mean shoot width} \times \text{mean density of shoot}/\text{m}^2$$

LAI is potentially more sensitive to environmental stress than is a parameter such as leaf width since it integrates both density and area (Neckles, 1994).

Shoot Biomass

Mean shoot biomass (dry weight of plant material per unit area) estimates are commonly used to assess the productivity of eelgrass beds and detect changes over time. The technique is universally accepted, however it requires destructive sampling and equipment that may not be available in all regions (ovens and scales).

Water Quality

The physical properties of seawater, especially in estuarine environments, fluctuate constantly in response to tides, currents, and volume of fresh water inflow. Many eelgrass monitoring programs incorporate environmental parameters into their study to provide a 'snapshot' of conditions that may, in turn, provide clues to significant water quality differences (Sewell, 2001).

The environmental parameters that are included in several large scale eelgrass monitoring projects are listed in Table 4. A brief summary of each program is provided in Appendix 1.

Table 4. Environmental variables included in several large scale eelgrass monitoring projects.

Parameter	**Puget Sound Submerged Vegetation Monitoring Project**	**SeagrassNet**	**European Directorate Special Areas of Conservation Program**
Temperature	√	√	-
Salinity	√	√	-
Dissolved oxygen	√		-
Turbidity	√	√	√
Photosynthetically Active Radiation	√	-	-
Light parameters, back scatter, florescence	√	-	-
Surface sediment character	-	√	-
Nutrient Levels	-	-	√

4.0 Strategy

The following strategy integrates four levels of study to enable all interested parties to participate in a large scale mapping effort. The level of detail that is selected to map and/or monitor an eelgrass meadow will be dependant on the specific goal of the study and the resources available. The use of standardized data dictionaries and data sheets ensures that all of the data that are collected are useful and may be integrated into the interactive database and mapping website (www.shim.bc.ca/maps.html).

The goals associated with each of the four levels, and a list of data required to achieve these goals are summarized below. The set parameters that must be assessed in order to meet the data requirements associated with each level are listed in Table 5. Details relating to the requirements are provided in Section 5.

Level 1

Goal: Conservation of intertidal eelgrass habitat

Requirements:
- identify the location of intertidal eelgrass meadows
- characterize the habitat within the intertidal area of the meadow

Level 2

Goal: Conservation of intertidal and subtidal eelgrass habitat

Requirements:
- identify the location and area of all eelgrass meadows
- characterize the habitat within the entire meadow

Level 3

Goal: Conservation of eelgrass meadows and early identification of habitat degradation or loss

Requirements:
- identify the location and area of all eelgrass meadows
- monitor eelgrass meadows to detect changes

Level 4

Goal: Conservation of eelgrass habitat and early identification of habitat degradation or loss and environmental stressors

Requirements:
- identify the location and area of all eelgrass meadows
- monitor eelgrass meadows to detect changes
- monitor changes in the surrounding environment water quality

Table 5. Minimum parameters to be assessed for each Level.

Parameter	Level 1	Level 2	Level 3	Level 4
location of eelgrass meadows	√	√	√	√
overview of intertidal habitat	√	√	√	√
overview of subtidal habitat		√	√	√
delineation of meadow(s)		√	√	√
maximum and minimum depth			√	√
distribution (degree of patchiness)			√	√
shoot density, including sexual status			√	√
Leaf Area Index (LAI)			√	√
turbidity			√	√
salinity				√
Total Suspended Solids (TSS)				√
chlorophyll A				√

5.0 Methods

The following methods are based on protocols that have been employed to map and monitor eelgrass communities. The methods are provided to enable groups or agencies to map eelgrass in a consistent manner, and to contribute to a central database using a standardized data entry form.

Mapping exercises should be completed during the summer, this will minimize the amount of variation between beds that is due to seasonal change. Monitoring should also be conducted during the summer, although the frequency of monitoring will depend on the resources of the study team. Monitoring programs may collect data annually (summer), biannually (summer and winter), or seasonally. Multiyear monitoring programs should be designed to ensure that field surveys are conducted within two weeks of the calendar date (month and day) of the original monitoring.

There are a minimum set of parameters associated with each level, however any of the parameters from higher levels may be included a survey. For example, a group may elect to complete a Level 1 survey but decide to collect shoot density data for the intertidal area with the methods used for a Level 3 & 4 survey.

Strategies may be developed to suit the requirements of each sampling team by using combination of levels. A recommended strategy is to map all eelgrass within a geographical area at Level 2, and then to select several meadows of interest to monitor at Level 3 or 4 on a regular basis. The meadows that are selected for monitoring would be in areas of potential environmental concern and at least one that is in a relatively protected area to use as a reference site.

A list of the equipment required for each level of study is provided in Appendix 2. Safety considerations for working in intertidal and subtidal eelgrass beds are provided in Appendix 3. Appendix 4 provides a suggested list of steps to complete each level of survey. A field datasheet and a draft of the data entry form are included in Appendix 5.

Location of Eelgrass Beds – All Levels

The first step is to identify the location of local eelgrass beds. It may take several years to locate all of the beds within a specific geographical area; depending on the time and resources that a specific group or organization has to dedicate to the project.

There are many sources of information that may assist in identifying the location of eelgrass beds. Sources that should be reviewed include: Herring Spawn Maps, Airphotos, Orthophotos, and the Community Mapping Network website (www.shim.bc.ca/maps.html).

The locations of eelgrass beds may be identified through low tide surveys, community surveys, diver surveys, and/or the use of a towed underwater video camera.

A survey of the low intertidal, conducted during the lowest daytime tides of the year, may be used to identify the location of many local eelgrass meadows. A survey of this type would only detect meadows that extend into the intertidal and would not provide information on the location of meadows that are restricted to subtidal areas.

Information may be solicited from the community. Local residents can provide information on the general locations of beds, which can later be assessed by the study team. The Shorekeepers manual provides many suggestions for gathering information from the community (http://www.pac.dfo-mpo.gc.ca/sci/protocol/shorekeepers/Guide/default.htm).

Diver surveys of the entire coastline are impractical, but may be used in areas where subtidal eelgrass is suspected.

A towed underwater video system can be used effectively to detect eelgrass beds. Underwater cameras that feed information into an above water videorecorder are available for $300 (black & white) and $1000 (colour). It is suggested that the habitat around –2 m to –5 m (chart datum) be investigated, as most subtidal eelgrass beds will extend across this depth.

The boundary of an eelgrass bed may be difficult to establish. In some cases it is very distinct, yet often the density of shoots slowly decreases around the perimeter. In order to be consistent, the Puget Sound study decided that areas that supported a minimum density of one (1) shoot per m^2 would be included in the bed. It is recommended that we adopt the same criteria. The edge of the bed shall be defined as the point at which the density decreases below 1 $shoot \cdot m^{-2}$, beyond which it continues to decrease. In areas that support a patchy distribution of eelgrass, there may be distances of several metres between patches. In these areas the edge of the bed should be located at the outer edges of the first and last patch.

Preliminary testing suggests that a hand held GPS may be as accurate as a differential GPS for mapping eelgrass beds. The results obtained by using a hand held (Garmin GPS 12XL without differential) and a differential (Trimble Pathfinder Pro XR) GPS were compared in an intertidal area of Comox Harbour. The two types of GPSs provided results within 1 metre of each other. Bill Mather (Coast Guard, Bamfield) reports that he has found the accuracy of a hand held GPS to be consistently within 5 metres on the sea, and frequently within 1 metre. Handheld GPSs should only be used with 3D NAV available with the averaging function enabled for capturing point data. Track logs can be used effectively to walk perimeters of beds. The locations may be also be drawn on orthophotos, charts, cadastral maps, or TRIM sheets depending on the scales at which these products are locally available.

Overview of Intertidal Habitat – All Levels

The data form provides a series of fields and categories to describe each bed. The fields include form, distribution, density, and substrate type.

There are two basic forms of eelgrass beds in the Pacific Northwest; fringing beds that occur as relatively narrow bands usually on gentle slopes, and more expansive beds that cover large areas such as tidal flats.

The distribution of eelgrass within the bed will be recorded as either continuous or patchy. Patchy beds are those that contain isolated groups or patches of plants. Beds, which are not patchy, will be classified as continuous; a bed that has a few bare patches would rate the continuous classification. A graphic representation of each distribution type is provided in Appendix 6.

An estimate of the percent cover of eelgrass at low tide, according to the categories supplied on the datasheet, is required. If the cover varies significantly then the primary, secondary, and, if necessary, tertiary densities should be recorded. Similarly, the common substrates should be recorded in order of dominance. If more than one percent cover class or substrate type is present then the percentage that is occupied by each type should be recorded according to the categories provided on the datasheet. Appendix 7 provides additional detail relating to percent cover assessments.

Reference photographs of the exposed bed should be taken during each survey. The photographs should include a site view and several close up photos of the eelgrass. An object, such as a metre-stick or pencil should be included in each close-up photo to provide a scale reference. Photographs should be taken from similar locations during subsequent surveys.

Overview of Subtidal Habitat – Levels 2, 3, and 4

The data required to provide an overview of the subtidal habitat mirrors that required to describe intertidal habitat.

Bed Delineation – Levels 2, 3, and 4

A GPS is used to georeference the boundaries of the eelgrass bed and create a polygon, which may be used to determine the area covered by eelgrass. The boundaries of the bed may be determined using; an aquaviewer, a diver or snorkler with weighted floats (Appendix 8), or a towed underwater camera. The depth to which the aquaviewer may be used successfully would depend on the turbidity of the water and the depth range of the eelgrass at each location.

GPS readings should be recorded at roughly 15 metre intervals around the perimeter of the bed.

The rules for defining boundaries and describing the bed follow those provided for intertidal eelgrass meadows above.

A detailed protocol for using a GPS to map the perimeter of eelgrass beds will be included in a subsequent version of this manual.

Maximum & Minimum Depth – Levels 3 & 4

The maximum and minimum depths should be determined when the bed is submerged. Divers depth gauges may only be used if they are known to be accurate to +/- 0.2 metres. One of the preferred methods is to have a weight attached to the end of a metre tape, which is lowered to a diver at the deepest and most shallow edge of the eelgrass bed. The diver places the weight on the bottom then tugs three times to notify the assistant on the boat that the line is in place. The assistant checks to make sure that the line is taught and vertical then records the measurement.

It is important to record the exact time that the measurement is recorded so that the reading may be adjusted to chart datum. Tidal heights over time may be downloaded from many sources including http://tbone.biol.sc.edu/tide/sites_othernorth.html.

Distribution – Levels 3 & 4

The distribution and zonation of eelgrass within a bed must be assessed in order to select the appropriate method for estimating shoot density.

Distribution

The distribution of eelgrass within the bed may be described as either patchy or continuous. Patchy beds are those that contain isolated groups or patches of plants. Beds, which are not patchy, will be classified as continuous; a bed that contains bare patches surrounded by eelgrass would be classified as continuous. A graphic representation of each distribution is provided in Appendix 6.

Zonation

The density and leaf size of eelgrass may be consistent throughout the bed, or may vary with depth. Typically, there are two or three zones within the bed, each located along a slightly different depth gradient. Each zone blends over several metres into the next; these areas are referred to as transition areas. The density and size of the shoots is significantly different between zones, therefore each zone must be sampled individually. Sampling should be conducted outside of the transition areas. The zones should be classified numerically starting with the uppermost zone. Zones that are less than 4 metres in width do not need to be assessed. The width of each zone does not need to be recorded as the exact boundaries are difficult, if not impossible to determine.

It is necessary to determine the number of zones within a bed in order to establish the number and location of transects to be sampled.

The following hypothetical description of an eelgrass bed is intended to provide the reader with an understanding the zonation typical in British Columbia.

> Zone 1 is a narrow band 8 metres wide, located in the low intertidal and shallow subtidal. The zone is characterized by a sparse population of short eelgrass (length 25 cm, density 32 shoots/m^2). Zone 1 blends into Zone 2, at a slightly lower elevation. The plants in Zone 2 are larger and more dense (80 cm, 112 shoots/m^2) than in those located in Zone 1. Zone 2 is 50 metres in width. The majority of the bed is located in Zone 2. Zone 2 merges into a third zone of sparse but larger plants (160 cm, 20 shoots/m^2) as the depth increases. Zone 3 is 10 metres wide.

Shoot Density

The protocol for density was designed to measure the mean density of shoots within the vegetated areas of the bed. Shoot density needs to be quantified within each zone. A 0.25 m^2 quadrat (50cm x 50 cm) should be used to assess density in most cases. This represents ¼ of a m^2.

Intertidal eelgrass may reach densities in excess of 500 shoots$\cdot$0.25m^{-2}. It is recommended that a smaller quadrat (25cm x 25 cm) be used to monitor density once the number of shoots$\cdot$0.25m^{-2} exceeds 100. A quadrat of this size represents 1/16 of a m^2.

Continuous Eelgrass Meadows

A temporary transect using a metre tape or marked line should be established in each zone, roughly parallel to the shore, along a depth continuum. The length of each transect should be roughly 60% of the bed width, to a maximum of 60 metres. The transects should be centred in the bed to avoid edge effects.

Predetermined random numbers will establish the location along either side of the transect where quadrats should be placed. Initially, thirty quadrats should be assessed for density within each zone. It will be necessary to determine the number of replicates (quadrats) that are required to estimate the mean density of shoots on a site specific basis due to the natural variability within eelgrass communities. The accepted method by which to accomplish this is to plot the running mean. Sample size is adequate once the variation between samples, which decreases as the number of samples increases, is reduced to 5%. It is likely that the number of replicates required will be less, however this number of samples should be sufficient to determine the running mean.

The total number of shoots rooted in each quadrat should be recorded, along with the total number of reproductive shoots in each quadrat. The number of vegetative shoots is calculated by subtracting the number of reproductive shoots from the total number of shoots.

Patchy Eelgrass Beds

It is challenging to design a sampling method for patchy (fragmented) beds as the size and distribution of patches will vary between and within sites. The following method may require revision.

Establish a temporary transect line parallel to shore. Start at the zero metre mark and record the length along the transect that is occupied by the first patch located under the transect line. If the area of the patch exceeds 1m^2, use a quadrat to determine the density (total number of shoots rooted within the quadrat and number of reproductive shoots) within 0.25m^2, avoiding the edges of the patch. If the patch is greater than 6m^2, monitor two quadrats within the patch. Attempts should be made to sample randomly, one method is to hover over the patch and allow the quadrat to drop to the bottom, and sample wherever it lands. Follow the transect line recording the distance that it travels over each patch, the distance between each patch, and the density within patches >1m^2.

Leaf Area Index (LAI)- Level 3 and 4

The mean leaf length and width can be determined from a random sample of 30 shoots. The data may be collected at the same time as the density is assessed. In order to avoid sampling only the largest shoots, measure the shoot located nearest to the upper right corner and the lower left corner of the quadrat. Measure the leaf length from sheath to tip of the second oldest leaf and the width near the middle of the leaf.

Calculate the LAI according to the following formula:

$$\text{LAI} = \text{mean shoot length} \times \text{mean shoot width} \times \text{mean density of shoot} / \text{m}^2$$

There are variations in the way that researchers measure LAI; some include the sheath, and others measure each leaf. The above method was selected, as it requires the least amount of time to calculate and can be used to provide a relative estimate of biomass.

Turbidity - Level 3 and 4

A secchi depth reading is recommended to assess turbidity.

Salinity - Level 4

A salinometer should be used to determine salinity, in parts per thousand (ppt).

Total Suspended Solids - Level 4

Water samples should be collected and taken to a local laboratory for analysis. The laboratory will provide a specific protocol for collecting and storing the samples.

Chlorophyll A - Level 4

Water samples should be collected and taken to a local laboratory for analysis. The laboratory will provide a specific protocol for collecting and storing the samples.

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Appendix 1 – Summary of Several Seagrass Mapping and Monitoring Programs

The following pages summarize several seagrass mapping and monitoring programs that have been recently implemented. Additional information may be obtained from the website addresses for each program.

Puget Sound Submerged Vegetation Monitoring Program

The objective of the Submerged Vegetation Monitoring Program is to 'quantify the state resource and its change over time' (Sewell et al., 2001). The four goals established by the program are:

1. Capture Temporal Trends in Eelgrass Distribution and Abundance in Puget Sound
2. Summarize Temporal Trends over Puget Sound and subareas
3. Monitor vegetation parameters that are strong indicators of eelgrass extent and quality
4. Link stressors to abundance and distribution. Six "core" sites will be sampled each year, and the remainder of Puget Sound will be sampled using rotational random sampling with partial replacement.

The program reviewed the available methodologies suited to goal 1 and selected linear transect sampling using a towed underwater video. Details are available in Norris et al., 2001a.

Methods that were considered and rejected included airborne remote sensing and colour air photo interpretation. Airborne remote sensing was rejected as the accuracy associated with this technique is +/- 40 feet which would not permit trend analysis, many of the beds in Puget Sound are located on beaches <40 feet wide, and the deep edge of many beds would not be visible. NOAA recommends using colour air photo interpretation, and stresses the importance of filming under optimal conditions, which are not always available in the Pacific Northwest.

SeagrassNet

*SeagrassNet is global monitoring program to investigate and document the status of seagrass resources world wide and the threats to this important and imperilled marine ecosystem (*www.seagrassnet.org*)*. The objectives of the program are to preserve seagrass ecosystems by increasing scientific knowledge and public awareness of this threatened coastal resource. The program began with seven countries in the Western Pacific and is expanding. The program uses a globally applicable monitoring protocol and a web-based interactive database. Each site is monitored on a quarterly basis.

The protocol involves determining distribution (including maximum and minimum depth), species composition, and abundance (cover, canopy height, shoot density (reproductive status) and above and below ground biomass) along permanent transects (parallel and perpendicular to the shore).

Environmental data is collected as follows:

water temperature - continuous reading at deep and shallow stations using tidbit data loggers,

light levels - % surface light using a Hobo light sensor, meters record data for two weeks at the time of each quarterly sampling, plus one land-based meter at a nearby location without shade,

salinity - water samples collected from three stations and analysed on a refractometer at a laboratory

surface sediment characteristics – estimates of the sediment type at three points on each cross transect and collect a core at each station on the primary transects

European Union Special Areas of Conservation

The European Union's Habitat Directive and developments to the Oslo and Paris Convention (OSPAR) lead to the creation of the Special Areas of Conservation (SAC) program. Eelgrass beds were identified as one of the habitats of major importance. Experts from academic and research institutes and nature conservation bodies compiled an Overview of Dynamics and Sensitivity Characteristics for Conservation Management of Zostera Biotopes. The review provides recommendations for mapping and monitoring.

The review states that "of the various monitoring techniques, airborne or sublittoral remote sensing (including side scan sonar) can rapidly map the distribution of beds over a large area, but must be ground-truthed by some other method. Underwater video and field observers (diving or shore) must be used to provide information on plant condition and associated biological community."

The review recommends the following parameters need to be monitored to detect change in the extent or health of eelgrass communities;

- ➢ distribution and extent of eelgrass coverage
- ➢ standing crop (biomass) and shoot density
- ➢ condition of shoots (leaf length, sexual status)
- ➢ occurrence of characteristic and representative species in the associated community
- ➢ local water quality (turbidity, nutrient levels)

Details are available at http://www.english-nature.org.uk/uk-marine/

Appendix 2 – Equipment

The following table lists the basic equipment that is required for each level of survey.

Equipment	Level 1	Level 2	Level 3	Level 4
Eelgrass Field Datasheets	√	√	√	√
maps or orthophotos at an appropriate scale, tidetables	√	√	√	√
boat (motor or paddle)		√	√	√
GPS		√	√	√
50 or 100 metre measuring tape or line			√	√
50 cm x 50 cm quadrats			√	√
metre stick			√	√
secchi disk			√	√
salinometer			√	√
Dive gear, snorkel gear, aquaviewer, or underwater camera			√	√
water quality sampling equipment				√

Waterproof notebooks or paper are highly recommended; these are available from stores that sell surveying equipment and some marine supply shops.

Quadrats may be constructed from any waterproof material. Local metal shops can usually make them out of aluminium for about $30. Aluminium quadrats are formed by a thin piece of 1” wide metal 2 metres in length that is bent to form a square and welded. Aluminium quadrats are recommended, as they are durable, rust proof, and are negatively buoyant so that they will lie flat on the substrate even if it is covered by water. Quadrats may also be made from wood or plastic pipe, although these types are more cumbersome to use and have a tendency to float.

A plastic coated surveyors measuring tape works well for marking transects. Alternatively, a thick nylon rope with labelled flagging tape to mark each metre may be constructed. The nylon tape has a tendency to float, this can be remedied by inserting short (e.g. 1” lengths) of lead wire into the rope at one metre intervals.

Secchi disks are used to measure the distance that one can see into the water, and to provide an indication of the turbidity. A secchi disk is a round flat disk, usually about 12” in diameter, with a cord attached in the centre. The surface of the disk is divided into four equal sized pie shaped triangles. The triangles are coloured white and black alternatively. The disk is lowered into the water and the depth at which it is no longer possible to distinguish the black from the white is recorded. A secchi disk may be purchased from a scientific supply company or hand made.

Tidetables are recommended to assist with planning the survey. Tidetables may be downloaded from http://tbone.biol.sc.edu/tide/sites_othernorth.html.

Appendix 3 – Safety Considerations

Intertidal Safety

The intertidal is a relatively safe place to work, however one should always be aware of the potential for injury. The most common cause of injury while working in and around intertidal eelgrass beds is from walking. Rocks and even mud, when covered with algae may be slippery. Rip rap (blasted rock that is often used as shore protection and to construct breakwaters), may be unstable; be cautious when climbing over it. People are often tempted to walk barefoot in soft glutinous mud, rather than loose their boots. However, broken shell embedded in the substrate can be sharp and may cut bare feet. Neoprene booties or old running shoes (with socks because the sand chaffs) work well.

Field work needs to be planned around the tides. On days when the low tide is less than 1 metre you can usually start work 1.5 to 2.0 hours before low tide, and continue for an hour afterwards. These times vary with other factors such as wind. If you are working around a headland, be sure to watch the tide; your return access may become blocked after the tide turns.

Never work alone, and carry a cellular phone or VHF radio in case of emergency. If possible try to include one member in each crew who has first aid certification. Always carry a first aid kit.

Bears and cougars frequent the backshore and sometimes intertidal areas in remote locations, so stay alert and keep an eye on the backshore for visitors.

It is a good idea to carry drinking water, as fecal coliform contamination and beaver fever is common in many of British Columbia’s streams and rivers.

Subtidal Safety

Boating

Safety regulations are available from the Canadian Coast Guard (www.ccg-gcc.cg.ca). The Coast Guard is phasing in operator requirements over several years. Currently, anyone born after April 1, 1983 must have a ‘proof of competency’ licence to legally operate a power boat. After September 15, 2002, anyone operating a power boat less than 4 meters in length must have a licence.

The safety regulations vary with size and type of boat. Boats (pleasure craft) less than six metres in length must be equipped with at least one personal floatation device for each person on board. Small motorized boats must also carry a paddle in case of engine failure or an anchor with 15 metres of rope, a bailer or manual pump, a 15 metre heaving line, a watertight flashlight or three flares, a sound signalling device (whistle or air horn), and navigation lights after sunset.

A basic boating safety course is available free of charge, on line at http://www.boatsafe.com/

SCUBA

Anyone participating in a SCUBA survey must be certified. A dive flag must be readily visible to warn boaters that divers are in the water. PADI recommends that a dive master be in attendance whenever a diver is in the water. The Reefkeepers manual has a section on diving safety that is available on line at http://www.pac.dfo-mpo.gc.ca/sci/protocol/reefkeepers/Guide/default.htm. Divers and boat operators must be aware of each other’s actions, and the danger associated with spinning propellers.

Appendix 4 – Project Planning

The following information is provided as a guide to assist with planning and organizing a field survey. Individual groups and organizations may want to modify the plan depending on the number of people available to assist with the survey.

The first step is to gather the background information (see Section 5 – Location of eelgrass beds) and review tide tables to select the best days for field work.

Level 1 Survey

1. Habitat Overview. Arrive on site within approximately 1 hour of low tide. Walk around the perimeter of the bed, then through it with the datasheet, thinking about the form, distribution, percent cover of eelgrass, and main substrate types in the bed. Avoid having many people follow the same path as excessive trampling can kill the eelgrass. Complete the Eelgrass Field Data Sheet – Section 1.
2. Georeference. Identify and map the edges of the bed with a GPS or on a map, airphoto, orthophoto, or chart.
3. Take photographs.

Tasks 1, 2, and 3 may be completed concurrently if the study team has enough members. The time required to complete a Level 1 survey will depend on the size of the study team and the area of the bed. A two member team could complete a Level 1 survey of a bed 100 metres wide or less within an hour.

Level 2 Survey

Intertidal areas of eelgrass beds should be surveyed at low tide as it will be much easier to assess them. Subtidal areas may be surveyed at any time, however the habitat may be easier to see if working from a boat, when there is less water at low tide.

1. Map the perimeter. It is always important to get a ‘big picture’ of the bed before you start the survey, either from a boat or underwater with SCUBA. Once the team has a fairly good idea as to the location of the bed, they can start mapping the perimeter.
2. Complete the Eelgrass Field Datasheet- Sections 1 and 2. In order to complete the datasheet, either the boat or divers will need to travel slowly over the bed, back and forth, until they feel that they have seen enough to complete the datasheet (habitat overviews). If possible, survey the intertidal area during low tide.

The perimeter mapping and habitat overviews may be completed simultaneously if there are adequate resources (boats and/or divers). It is estimated that one hour will be required to map the perimeter, and one hour to assess the habitat.

Level 3 Survey

Intertidal areas of eelgrass beds should be surveyed at low tide as it will be much easier to assess them. Subtidal areas may be surveyed at any time, however the habitat may be easier to see from a boat when there is less water at low tide. The entire survey does not need to be completed in one day, however it should be completed within one calendar week.

1. Map the perimeter. It is always important to get a ‘big picture’ of the bed before you start the survey, either from a boat or underwater with SCUBA. Once the team has a fairly good idea as to the location of the bed, they can start mapping the perimeter.

2. Complete the Eelgrass Field Datasheet – Sections 1 and 2. In order to complete the datasheet, either the boat or divers will need to travel slowly over the bed, back and forth, until they feel that they have seen enough to complete the datasheet (habitat overviews). If possible survey the intertidal area during low tide.
3. Determine maximum and minimum depths.
4. Determine the number of zones and select locations for transects.
5. Establish transects, collect shoot density data, and leaf length and width data.
6. Secchi depth reading may be taken at any time during the survey.

Study teams that include more than one pair of divers may decide to dedicate one team to mapping the perimeter and determining maximum and minimum depths, while the other pair(s) complete tasks 4 and 5.

Calculations (means, leaf area indices) may be completed subsequent to the field survey.

A study team of one boat tender and two divers would require approximately 5 hours to complete the survey. A study team of one boat tender, two teams of divers, and two people to assess the intertidal could complete the survey in less than 2 hours.

Level 4 Survey

Refer to the instructions for a Level 3 Survey. Collect water samples at any time, but remember to record the time of collection on the datasheet.

Appendix 5 – Field Data Form & Data Entry Form

A field data form (p. 22 – 27) and images of the electronic data entry forms are provided (p. 28-32). The 'Eelgrass Field Data Sheet' may be photocopied onto waterproof paper for use during fieldwork. The 'Eelgrass Bed Mapping Data Entry Form' (EBMDEF) is a snapshot of the one that can be used to enter data into the interactive web based database. In order to enter data into the Community Mapping Network database (http://www.shim.bc.ca/eelgrass/main.htm) each group will be assigned a username and password. The data from the field data sheet may then be submitted electronically. A help menu is available on the toolbar.

Eelgrass Field Data Sheet

Background

Location: ....

Date: ....*(dd/mm/yr)*

Primary Field Surveyor:...

Crew: ....

....

Time start:.....................................   Time finish: ....

Tide height start: ....   Tide height finish: ....

Level of Survey: ....   Tidal range of eelgrass bed (subtidal, intertidal, both):………………..

Platform used to survey eelgrass bed (shore, boat, dive, video): ....

Reference used to determine tide height: ....

Reference map type:………………………………………………………………………………………………….

Reference map name or number:…………………………………………………………………………………

Reference map scale:………………………………………………………………………………………………

Geographic (Lat./Long.) or Projection:……………………………………………………………………………

Specifics of Projection (UTM, Albers, etc. including zone and other details):……………………………………………..

Method and Level of accuracy to which bed was mapped (circle one)

Code	Map Accuracy
1	Location measured using GPS (see GPS model and accuracy fields)
2	Location generalized from DFO log book lat/long positions
3	Location indicated to 2 mm at chart scale
4	Alongshore location indicated to 2mm at chart scale; across shore accuracy unknown
5	General location only; rough sketch on chart or place name (5 mm at chart scale)
6	Tied to shoreunit or other shoreline segment
7	Tied to DFO Statistical Subarea
8	Tied to DFO Statistical Area
9	Alongshore location indicated to 5 mm at chart scale, across shore accuracy unknown
10	Vague location only (1-2 cm at chart scale)

Method used to georeference (GPS/hardcopy map/orthophoto/airphoto): ....

Make and Model of GPS (if one was used):………………………………………………………………………………..

Comments specific to the eelgrass bed (health, adjacent backshore land use, backshore structures, possible threats)

1. Overview of Intertidal Habitat: All Levels – if bed is restricted to the subtidal go to Section 2.

Form Fringing ☐ Flat ☐
Distribution Continuous ☐ Patchy ☐

Percent Cover of intertidal eelgrass

Primary	1 to 10%	....	Secondary	1 to 10%	....	Tertiary	1 to 10%	....
	11 to 25	....	(optional)	11 to 25%	....	(optional)	11 to 25%	....
	26 to 50%	....		26 to 50%	....		26 to 50%	....
	51 to 75%	....		51 to 75%	....		51 to 75%	....
	> 75%	....		> 75%	....		> 75%	....

Substrate Type

Primary	mud	....	Secondary	mud	....	Tertiary	mud	....
	mud/sand	....	(optional)	mud/sand	....	(optional)	mud/sand	....
	sand	....		sand	....		sand	....
	gravel	....		gravel	....		gravel	....
	cobble	....		cobble	....		cobble	....
	boulder	....		boulder	....		boulder	....
	bedrock	....		bedrock	....		bedrock	....

2. Overview of Subtidal Habitat: Levels 2, 3, and 4

Form Fringing ☐ Flat ☐
Distribution Continuous ☐ Patchy ☐

Percent Cover of subtidal eelgrass ( )

Primary	1 to 10%	....	Secondary	1 to 10%	....	Tertiary	1 to 10%	....
	11 to 25	....	(optional)	11 to 25%	....	(optional)	11 to 25%	....
	26 to 50%	....		26 to 50%	....		26 to 50%	....
	51 to 75%	....		51 to 75%	....		51 to 75%	....
	> 75%	....		> 75%	....		> 75%	....

Area occupied by: ( )

Primary	1 to 10%	....	Secondary	1 to 10%	....	Tertiary	1 to 10%	....
	11 to 25	....	(optional)	11 to 25%	....	(optional)	11 to 25%	....
	26 to 50%	....		26 to 50%	....		26 to 50%	....
	51 to 75%	....		51 to 75%	....		51 to 75%	....
	> 75%	....		> 75%	....		> 75%	....

Substrate Types ( )

Primary	mud	....	Secondary	mud	....	Tertiary	mud	....
	mud/sand	....	(optional)	mud/sand	....	(optional)	mud/sand	....
	sand	....		sand	....		sand	....
	gravel	....		gravel	....		gravel	....
	cobble	....		cobble	....		cobble	....
	boulder	....		boulder	....		boulder	....
	bedrock	....		bedrock	....		bedrock	....

Area occupied by ( )

Primary	1 to 10%	....	Secondary	1 to 10%	....	Tertiary	1 to 10%	....
	11 to 25	....	(optional)	11 to 25%	....	(optional)	11 to 25%	....
	26 to 50%	....		26 to 50%	....		26 to 50%	....
	51 to 75%	....		51 to 75%	....		51 to 75%	....
	> 75%	....		> 75%	....		> 75%	....

3. Depth: Levels 3 and 4

Method used to determine Maximum Depth
(diver with depth gauge, diver with boat and metre tape or rod, survey rod without diver, other –explain)

....
....
....
....
....

.... Time measurement was taken
.... Depth Reading (metres e.g. 8.2 m)
.... Tide height at this time
.... Actual depth

Method used to determine Maximum Depth
(diver with depth gauge, diver with boat and metre tape or rod, survey rod without diver, other –explain)

....
....
....
....
....

.... Time measurement was taken
.... Depth Reading (metres)
.... Tide height at this time
.... Actual depth

4. Distribution & Density: Levels 3 and 4

Distribution Continuous - *proceed to Section 4A*
Patchy - *proceed to Section 4B*

4A. Continuous Eelgrass – complete one form for each zone

Number of Zones:..................................

Zone #:					
length of transect			**# of quadrats sampled**		
raw data (#/0.25m^2)					
total	**reproductive**	**total**	**reproductive**	**total**	**reproductive**
mean # total:			**mean # reproductive:**		

4B. Patchy Eelgrass – complete one form for each zone

Number of Zones: ....
Direction of Transect (e.g. 0m at north end):...........................

Zone #: ....

Distance across eelgrass patch (e.g. 2.4m)	**# shoots / $0.25m^2$**	**Distance to next eelgrass patch**

Mean # shoots/$0.25m^2$ (within patches): ....

4. Leaf Area Index (LAI): Levels 3 and 4

sample	length	width
1.		
2.		
3.		
4.		
5.		
6.		
7.		
8.		
9.		
10.		
11.		
12.		
13.		
14.		
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16.		
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18.		
19.		
20.		
21.		
22.		
23.		
24.		
25.		
26.		
27.		
28.		
29.		
30.		
Ó (total)		
x̄ (Ó ÷ 30)		

Mean leaf length (x̄): ………… ....Mean leaf width (x̄): ……………

Leaf Area Index (mean leaf length x mean leaf width x mean shoot density): …………….

5. Turbidity: Levels 3 and 4

Turbidity (secchi depth reading): ....

Time that reading was taken: ....

6. Salinity, Total Suspended Solids, Chlorophyll A: Level 4

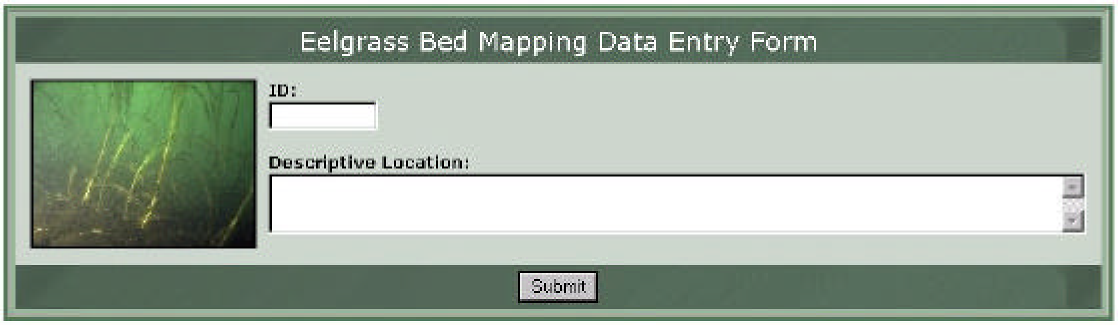
Salinity: ....

Total Suspended Solids: ....

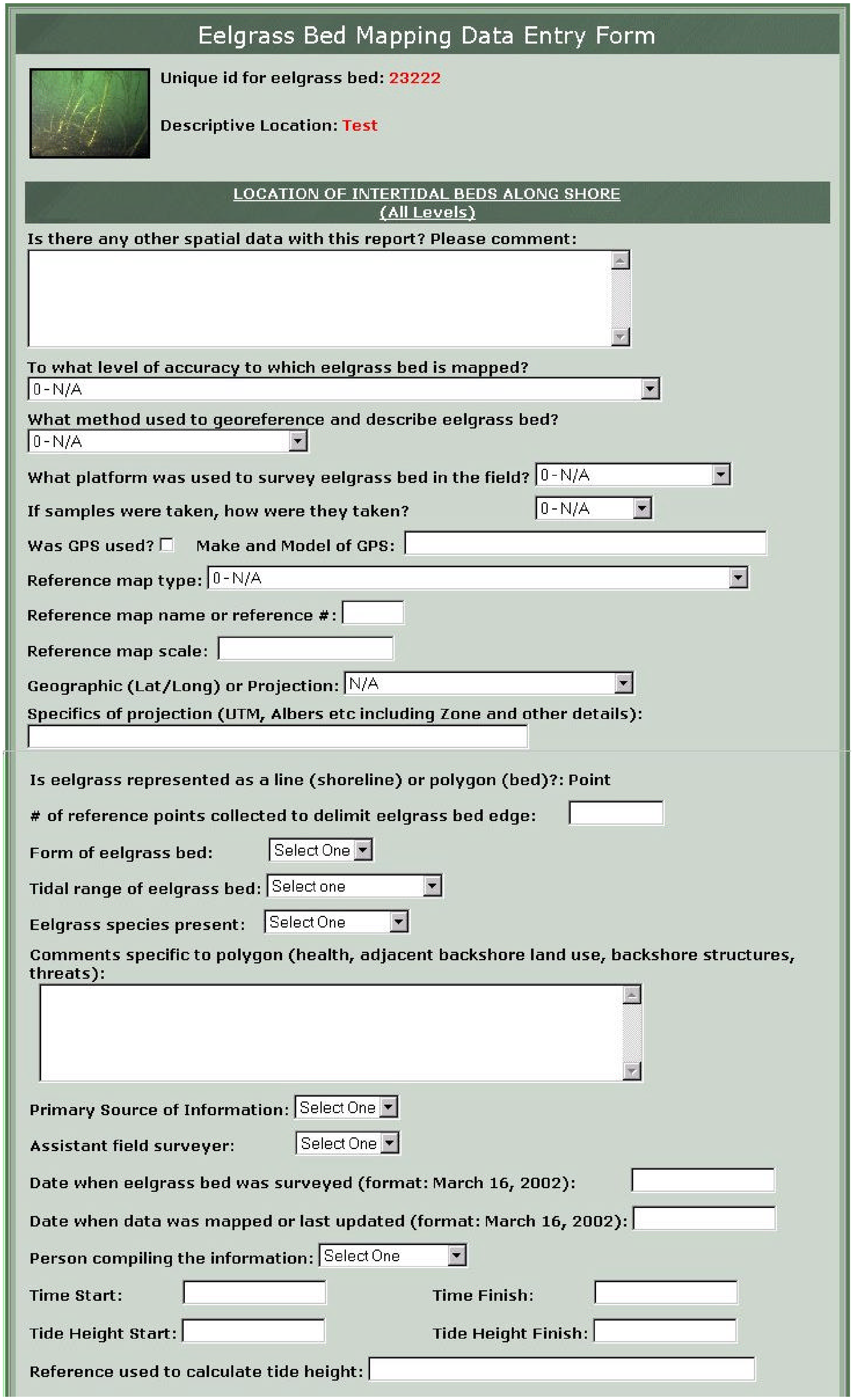
Chlorophyll A: ....

Time that samples were collected: ....

Form 1:



Form 2 (Main form):



Eelgrass Bed Mapping Data Entry Form

Unique id for eelgrass bed: 23222

Descriptive Location: Test

LOCATION OF INTERTIDAL BEDS ALONG SHORE
(All Levels)

Is there any other spatial data with this report? Please comment:

To what level of accuracy to which eelgrass bed is mapped? 0 - N/A

What method used to georeference and describe eelgrass bed? 0 - N/A

What platform was used to survey eelgrass bed in the field? 0 - N/A

If samples were taken, how were they taken? 0 - N/A

Was GPS used? ☐ Make and Model of GPS:

Reference map type: 0 - N/A

Reference map name or reference #:

Reference map scale:

Geographic (Lat/Long) or Projection: N/A

Specifics of projection (UTM, Albers etc including Zone and other details):

Is eelgrass represented as a line (shoreline) or polygon (bed)?: Point

# of reference points collected to delimit eelgrass bed edge:

Form of eelgrass bed: Select One

Tidal range of eelgrass bed: Select one

Eelgrass species present: Select One

Comments specific to polygon (health, adjacent backshore land use, backshore structures, threats):

Primary Source of Information: Select One

Assistant field surveyer: Select One

Date when eelgrass bed was surveyed (format: March 16, 2002):

Date when data was mapped or last updated (format: March 16, 2002):

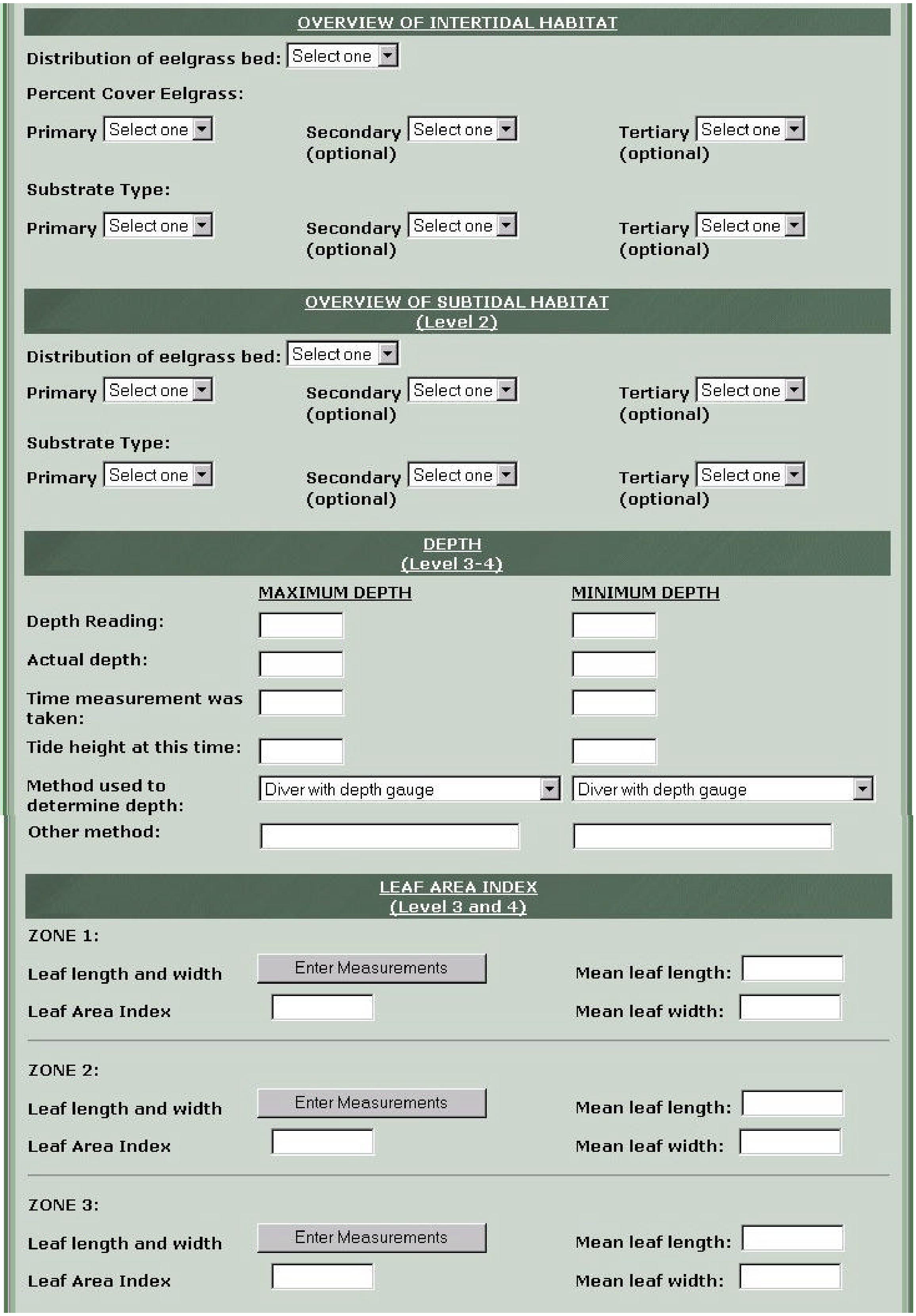
Person compiling the information: Select One

Time Start: Time Finish:

Tide Height Start: Tide Height Finish:

Reference used to calculate tide height:

Eelgrass Bed Data Entry Form



OVERVIEW OF INTERTIDAL HABITAT

Distribution of eelgrass bed: Select one

Percent Cover Eelgrass:

Primary Select one
Secondary (optional) Select one
Tertiary (optional) Select one

Substrate Type:

Primary Select one
Secondary (optional) Select one
Tertiary (optional) Select one

OVERVIEW OF SUBTIDAL HABITAT (Level 2)

Distribution of eelgrass bed: Select one

Primary Select one
Secondary (optional) Select one
Tertiary (optional) Select one

Substrate Type:

Primary Select one
Secondary (optional) Select one
Tertiary (optional) Select one

DEPTH (Level 3-4)

	MAXIMUM DEPTH	MINIMUM DEPTH
Depth Reading:		
Actual depth:		
Time measurement was taken:		
Tide height at this time:		
Method used to determine depth:	Diver with depth gauge	Diver with depth gauge
Other method:		

LEAF AREA INDEX (Level 3 and 4)

ZONE 1:

Leaf length and width Enter Measurements
Mean leaf length:
Leaf Area Index
Mean leaf width:

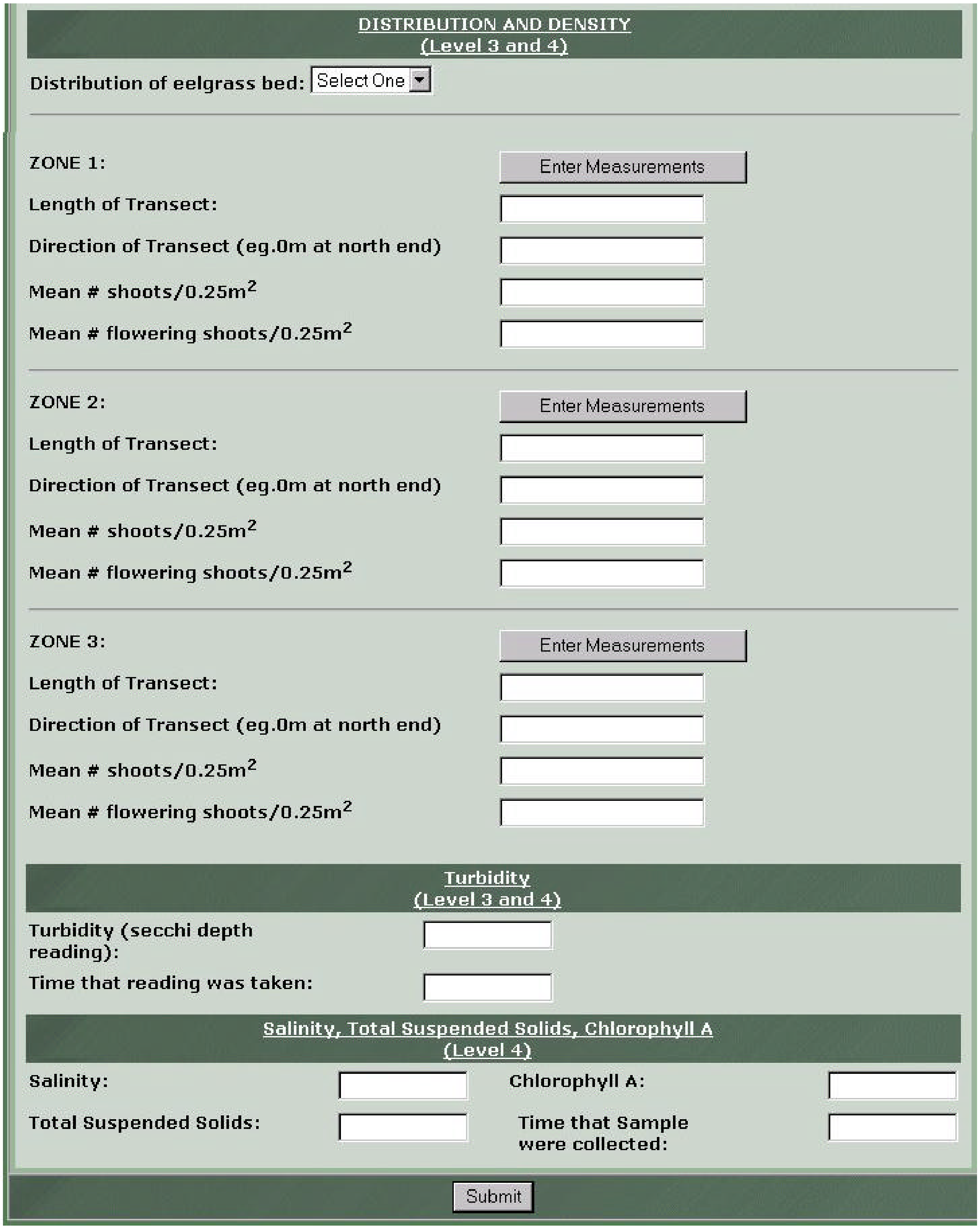
ZONE 2:

Leaf length and width Enter Measurements
Mean leaf length:
Leaf Area Index
Mean leaf width:

ZONE 3:

Leaf length and width Enter Measurements
Mean leaf length:
Leaf Area Index
Mean leaf width:

Eelgrass Bed Data Entry Form



DISTRIBUTION AND DENSITY
(Level 3 and 4)

Distribution of eelgrass bed: Select One

ZONE 1:
Length of Transect:
Direction of Transect (eg.0m at north end)
Mean # shoots/$0.25m^2$
Mean # flowering shoots/$0.25m^2$

ZONE 2:
Length of Transect:
Direction of Transect (eg.0m at north end)
Mean # shoots/$0.25m^2$
Mean # flowering shoots/$0.25m^2$

ZONE 3:
Length of Transect:
Direction of Transect (eg.0m at north end)
Mean # shoots/$0.25m^2$
Mean # flowering shoots/$0.25m^2$

Turbidity
(Level 3 and 4)

Turbidity (secchi depth reading):
Time that reading was taken:

Salinity, Total Suspended Solids, Chlorophyll A
(Level 4)

Salinity:
Chlorophyll A:
Total Suspended Solids:
Time that Sample were collected:

Eelgrass Bed Data Entry Form

LAI Form:

ZONE 1 - Leaf Area Index

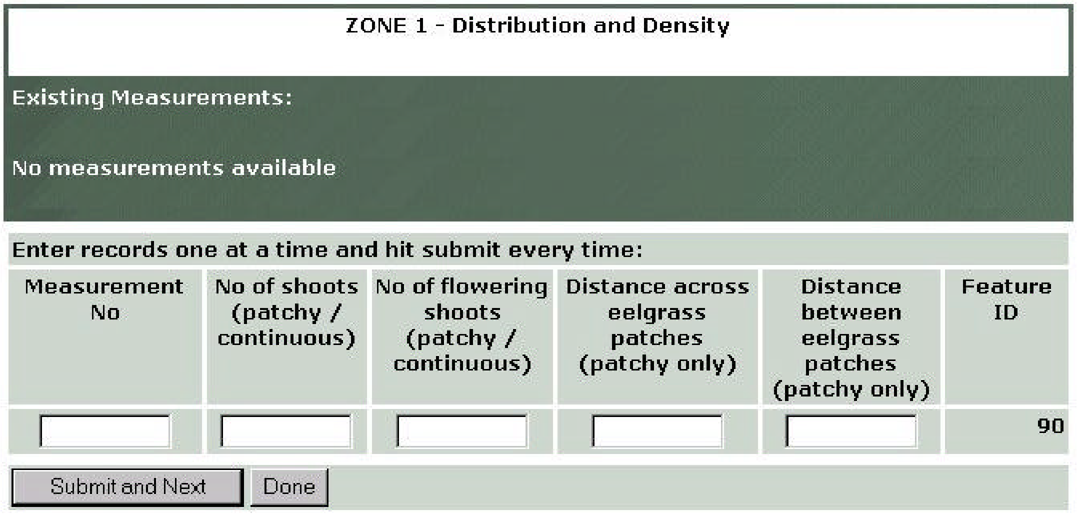
Existing Measurements:

No measurements available

Enter records one at a time and hit submit every time:

Measurement No	Width	Length	Feature ID
			90

Distribution and Density Form:



ZONE 1 - Distribution and Density

Existing Measurements:

No measurements available

Enter records one at a time and hit submit every time:

Measurement No	No of shoots (patchy / continuous)	No of flowering shoots (patchy / continuous)	Distance across eelgrass patches (patchy only)	Distance between eelgrass patches (patchy only)	Feature ID
					90

Appendix 6 – Patchy vs. Continuous Eelgrass Distribution

The following illustrations are provided to demonstrate the difference between patchy and continuous eelgrass cover. The term Continuous is used to indicate that eelgrass is distributed over most of the area within the bed (Figure 1). There may be some areas without eelgrass within the bed (Figure 2).

Eelgrass is described as patchy when the bed or meadow is composed of many patches or islands of eelgrass, most of which are surrounded by areas without eelgrass (Figure 3). The area between patches is usually either exposed substrate or macroalgae.

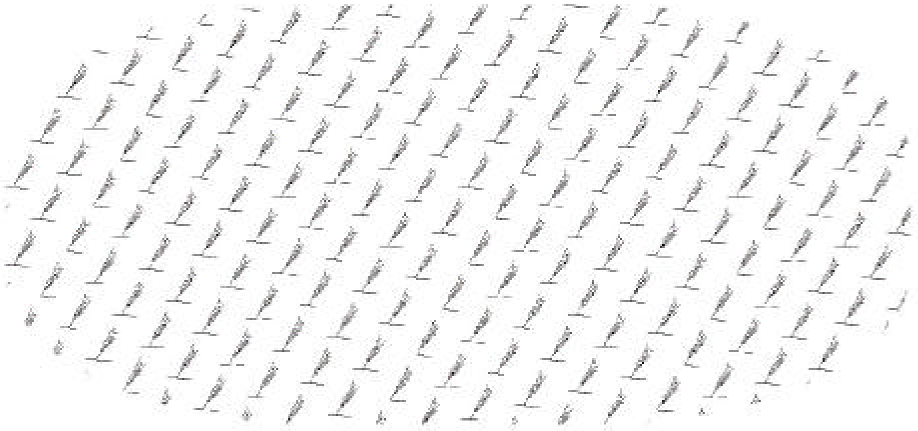


Figure 1. Continous Cover

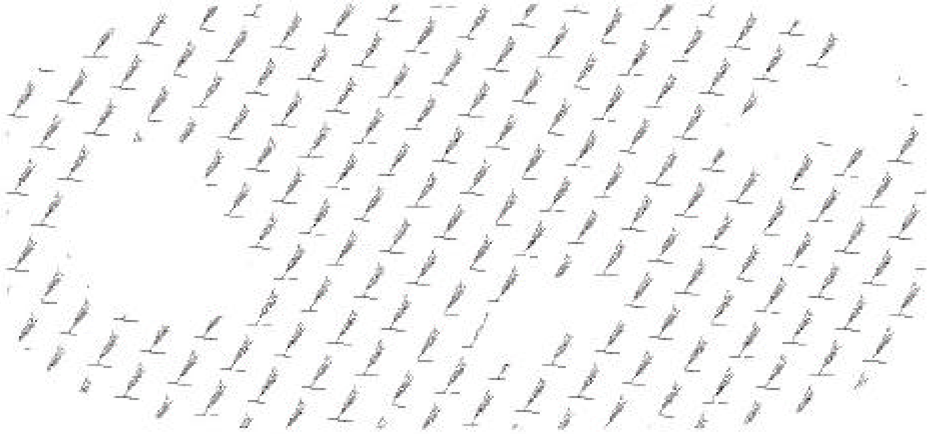


Figure 2. Continous cover with bare patches

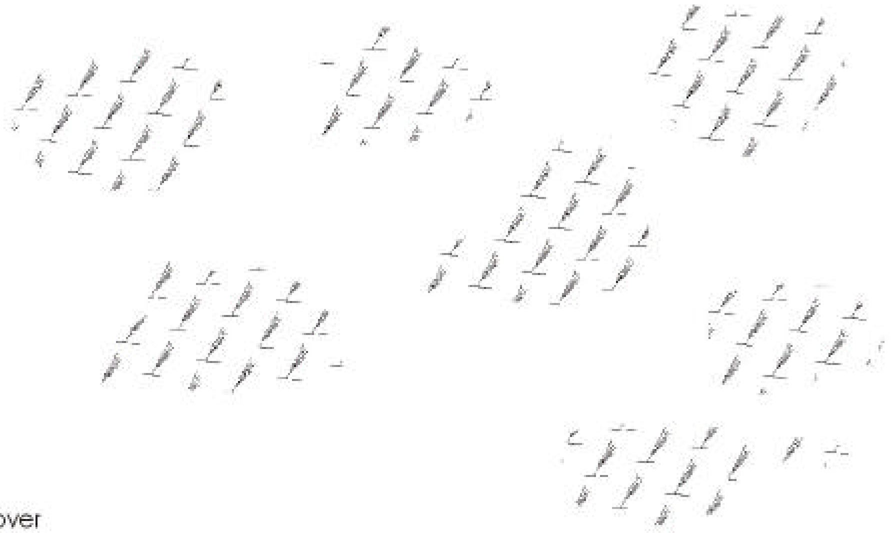


Figure 3. Patchy Cover

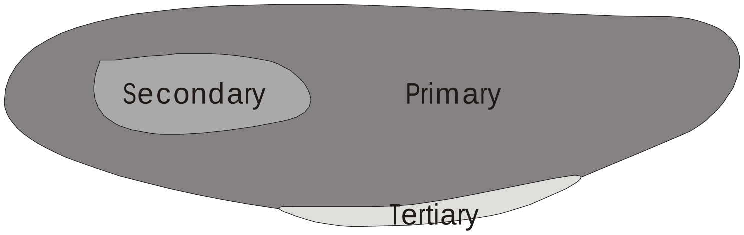
Appendix 7 – Percent Cover

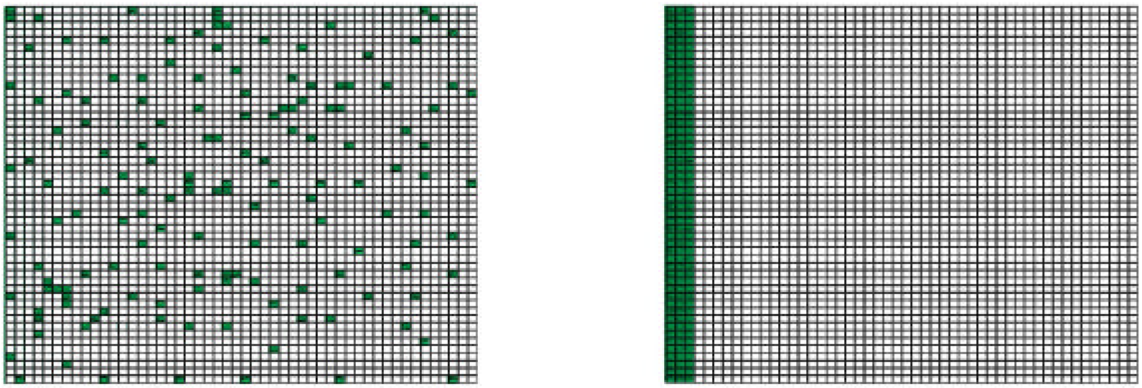
Percent cover is a quantitative assessment of the area covered by plants. For example, when the leaves and shoots form a dense blanket over the substrate (ground) such that it is impossible to see the substrate below the plants the percent cover is 100%. If you can see the substrate between the plants then the percent cover is less than 100%. The following figures are provided to illustrate this concept.

Imagine that the grey squares represent cover by eelgrass; the white squares represent exposed substrate (no eelgrass). Some people find it helpful to mentally move all the plants together in order to estimate the percent cover. Figure 7.1a represents a sparse eelgrass bed where only 6% of the area is covered by eelgrass. Figure 7.1b contains the same number of grey squares but they have been moved together. Accurately estimating precise percent cover requires training and experience. A way to circumvent this problem is to estimate percent cover within ranges. The datasheet provides a series of ranges that can be used to evaluate percent cover. By looking at the area covered by eelgrass, and perhaps mentally shifting all the plants together, you can determine which range best reflects the percent cover of eelgrass in the bed. For example, the diagram shown in Figure 7.1a would fall between 1% and 10%. The ranges that are used in this study are listed below.

Primary	1 to 10%	....	Secondary	1 to 10%	....	Tertiary	1 to 10%	....
	11 to 25	....	(optional)	11 to 25%	....	(optional)	11 to 25%	....
	26 to 50%	....		26 to 50%	....		26 to 50%	....
	51 to 75%	....		51 to 75%	....		51 to 75%	....
	> 75%	....		> 75%	....		> 75%	....

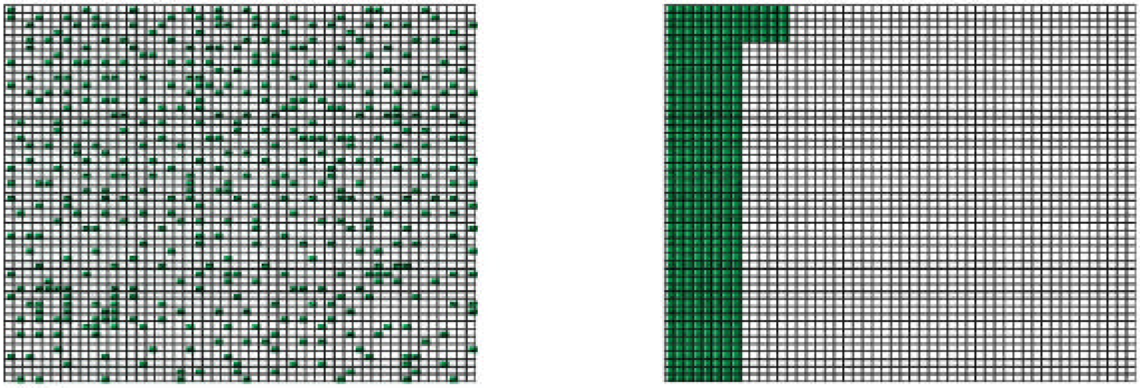
There are often differences in percent cover within a bed due to variations in physical variables such as depth or substrate. The following diagram provides a graphic representation of a bed that is composed of three areas with distinctly different percent covers. The dark area represent very dense eelgrass (>75%), the light area represents an area with low percent cover (1-10%), and the mid shade an area with intermediate cover (26 – 50%). Since most of the area falls into the >75% range this would be the primary percent cover. The secondary and tertiary percent covers would be 1-10% and 26-50% respectively. The secondary and tertiary percent cover estimates are considered optional as many beds are relatively uniform within the broad ranges that are provided. An area should represent at least 10% of the total area before it is considered significant enough to note on the datasheet.





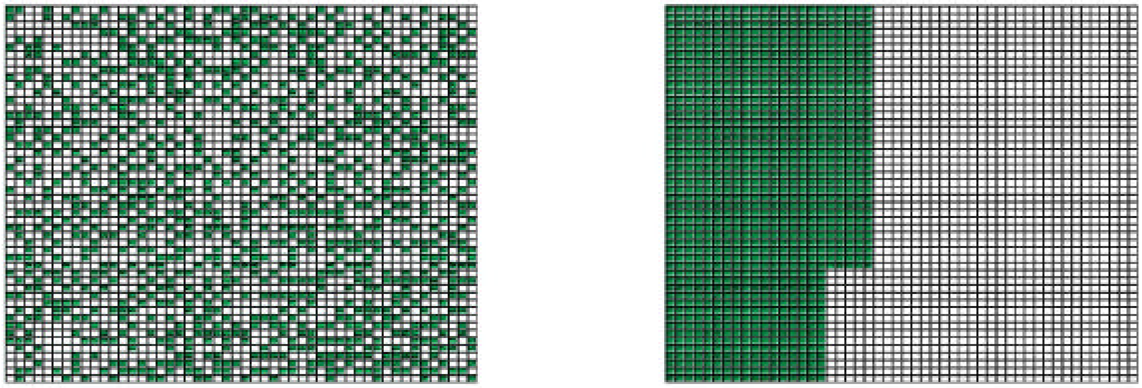
6%

Figure 7.1 Six percent of the squares are shaded. The squares are randomly located in the first diagram (a) and are grouped in the second (b). This represents an area that would be classified as 1-10% cover on the datasheet.



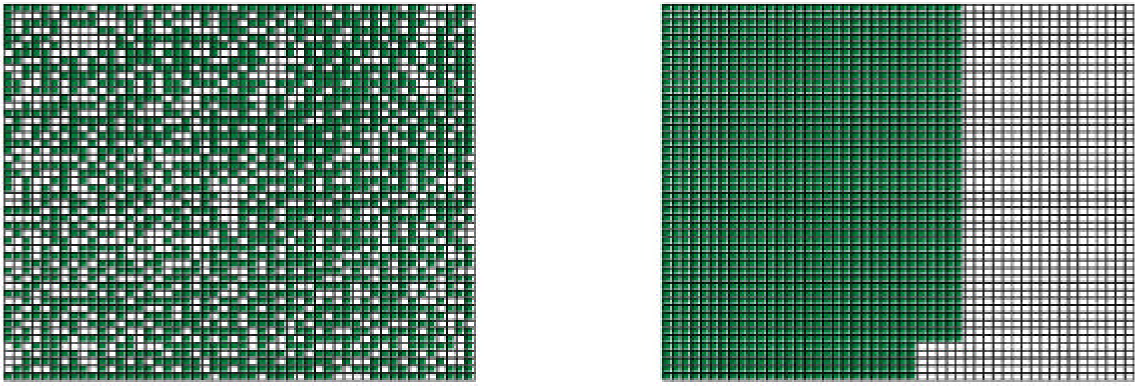
18%

Figure 7.2 Eighteen percent of the squares are shaded. The squares are randomly located in the first diagram (a) and are grouped in the second (b). This represents an area that would be classified as 11-25% cover on the datasheet.



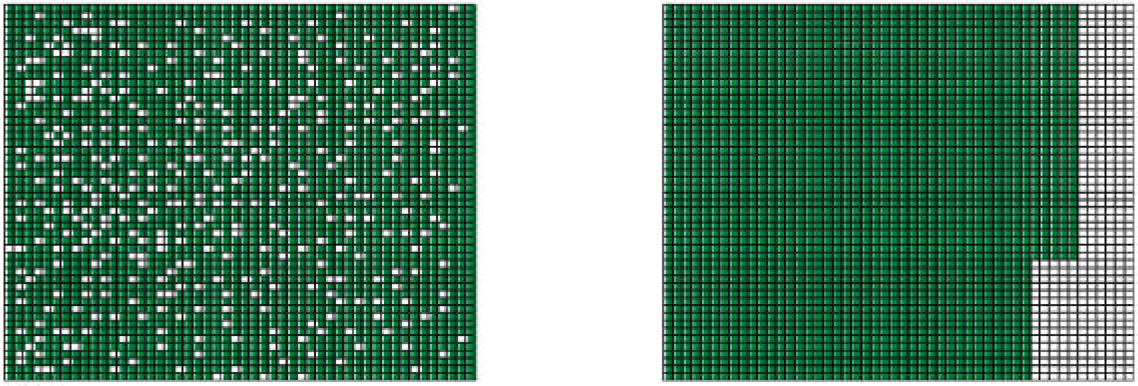
41%

Figure 7.3 Forty-one percent of the squares are shaded. The squares are randomly located in the first diagram (a) and are grouped in the second (b). This represents an area that would be classified as 26-50% cover on the datasheet.



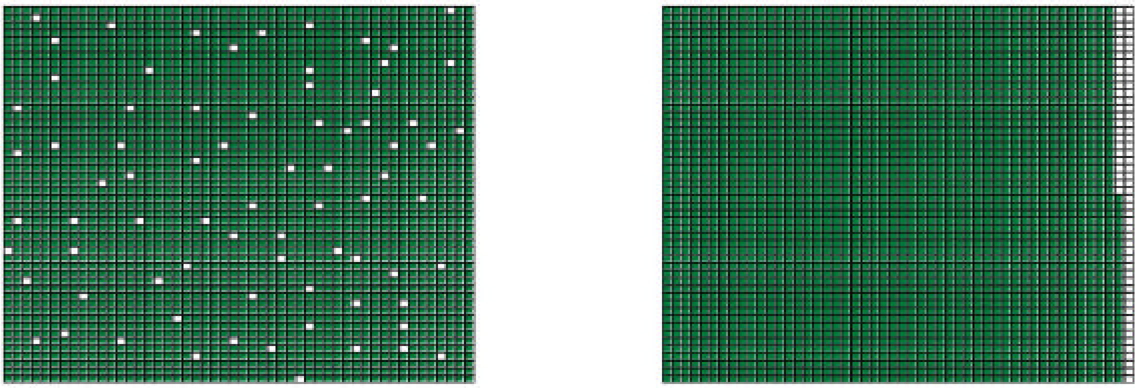
63%

Figure 7.4 Sixty-three percent of the squares are shaded. The squares are randomly located in the first diagram (a) and are grouped in the second (b). This represents an area that would be classified as 51-75% cover on the datasheet.



85%

Figure 7.5 Eighty-five percent of the squares are shaded. The squares are randomly located in the first diagram (a) and are grouped in the second (b). This represents an area that would be classified as >75% cover on the datasheet.



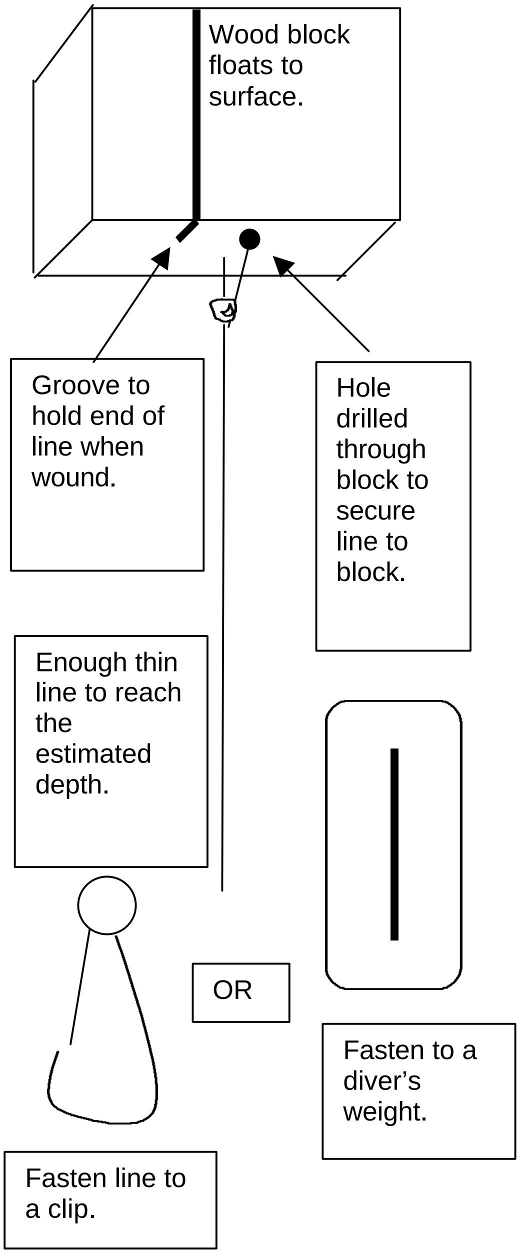
97%

Figure 7.6 Ninety-seven percent of the squares are shaded. The squares are randomly located in the first diagram (a) and are grouped in the second (b). This represents an area that would be classified as >75% cover on the datasheet.

Appendix 8 – Marker Floats

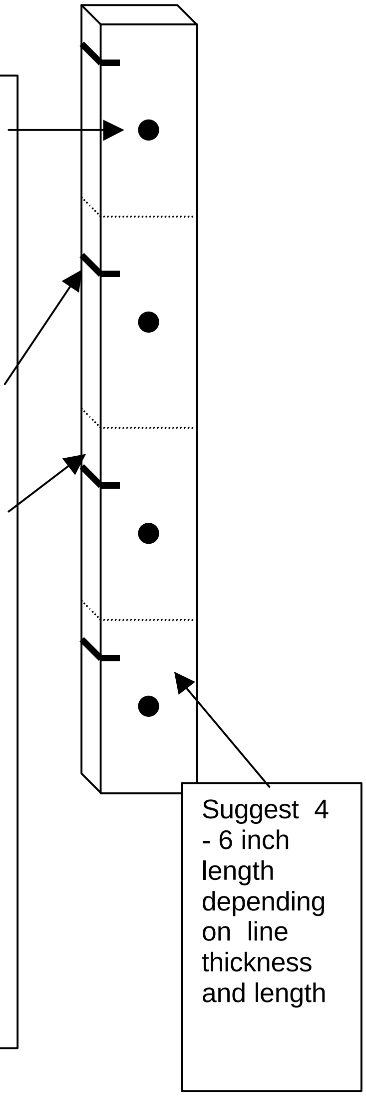
The following float design was developed by Sarah Verstegen of SeaChange to mark the perimeter of eelgrass beds.

If you need to mark the location of eelgrass under water so that you can find it from the surface, try these for short-term use. The line is wound around the block and notched into the groove. A diver can carry a few in a goody bag. When the diver finds a location to mark for people at the surface, she or he sets the marker weight on the bottom. (Clips work when there is something to fasten to.) Then, s/he un-notches the line from the groove. The line will unreel itself from the block as it floats to the surface. It helps divers avoid that nasty tangle of line when working under water.



Use either a 2 x 4 or 2 x 3 inch piece of lumber. It's easier to make notches in a long piece before it's cut into the smaller blocks. Plastic clips are ligher than lead weights and cheaper than brass.

1. Drill holes for line. Use a bit slightly larger than your line diameter.
2. Cut the grooves. Set your saw blade for the desired depth.
3. Cut each block from the length.
4. Paint blocks a bright color.
5. Number each one.
6. Thread line through hole and tie securely.
7. Wind line around block.
8. Tie end of line to chosen bottom piece.



Habitat Management Qualitative Risk Assessment : Water Column Oyster Aquaculture in New Brunswick

S. Bastien-Daigle, M. Hardy and G. Robichaud

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Moncton (New Brunswick) E1C 9B6

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Canadian Technical Report of Fisheries and Aquatic Sciences 2728

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Cat. No. Fs 97-6/2728E ISSN 0706-6457

Published by:

Oceans and Science Branch
Fisheries and Oceans
Gulf Region
P.O. Box 5030
Moncton (New Brunswick) E1C 9B6

Correct citation:

Bastien-Daigle, S., M. Hardy and G. Robichaud. 2007. Habitat Management Qualitative Risk Assessment: Water Column Oyster Aquaculture in New Brunswick. Can. Tech. Rep. Fish. Aquat. Sci. 2728 : vii + 72p.

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ABSTRACT

The Department of Fisheries and Oceans Canada (DFO) is responsible for evaluating potential environmental impacts on fish habitat associated with project development. Aquaculture of the native oyster (*Crassostrea virginica*) has been expanding in gulf New Brunswick's (N.B.) coastal communities, thus, a qualitative risk assessment was initiated. This involves an evaluation of water column oyster aquaculture and its interactions with fish habitat, as defined in the *Policy for the Management of Fish Habitat,* by integrating a thorough review of the current scientific information and a description of the oyster aquaculture industry. This assessment follows the work of the National Advisory Process which characterized the potential environmental risks of bivalve aquaculture in the marine environment. That scientific review is complemented with technical data as well as additional information to specifically characterize the potential effects of oyster aquaculture in N.B. The present qualitative risk assessment is intended to assist habitat managers in their decision-making process and is based on the *Habitat Management Program Risk Management Framework*. The framework provides a structured process for characterizing the potential risks and assessing their significance in regards to the productive capacity of fish habitat. An Ecological Risk Assessment and a Net Ecological Benefit Analysis are used to make determinations as to the effects and functions, respectively, of water column oyster aquaculture in gulf N.B. Using the risk assessment, we conclude that the overall "scale of potential negative effects" of water column oyster aquaculture and the "sensitivity of fish and fish habitat" correspond to a low-risk activity which is not likely to significantly harm the productive capacity or the ecological integrity of fish habitat. Moreover, our analysis suggests that oysters in aquaculture can potentially be of significant benefit to these estuaries and can help to restore many important ecological functions which were reduced following the historical decline of natural populations. Given the nature of this activity, we conclude that the risks associated with water column oyster aquaculture can be managed in a sustainable manner with adequate planning and mitigation measures through an adaptive management approach.

RÉSUMÉ

Le Ministère des Pêches et des Océans du Canada (MPO) est responsable d'évaluer les effets environnementaux potentiels des projets de développement sur l'habitat du poisson. L'aquaculture de l'huître indigène (*Crassostrea virginica*) est une activité en croissance au Nouveau-Brunswick (N.-B.). Pour cette raison, une évaluation qualitative du risque de cette activité a été entreprise. Une évaluation des interactions entre l'ostréiculture en colonne d'eau et l'habitat du poisson a été effectuée, tel que définie sous la Politique de gestion de l'habitat du poisson, par l'entremise d'une revue d'informations scientifiques et une description de l'activité ostréicole. Cette évaluation fait suite au processus officiel d'avis scientifique qui a caractérisé les risques environnementaux potentiels de la culture marine des bivalves. Cette revue scientifique ainsi que d'autres études et informations techniques ont été utilisées afin de caractériser plus spécifiquement les effets de l'ostréiculture dans la colonne d'eau au N.-B. L'évaluation qualitative du risque a comme objectif d'aider les gestionnaires dans le processus de prise de décisions selon le *Cadre de gestion de risques* du *Programme de gestion de l'habitat*. Ce cadre offre un processus structuré qui permet de définir les risques et déterminer leur importance en fonction de la capacité productive de l'habitat du poisson. Une évaluation du risque écologique et une analyse du bénéfice écologique net ont été utilisées afin de déterminer les effets et les fonctions, respectivement, de l'ostréiculture dans la colonne d'eau au N.-B. Cette analyse nous a permis de conclure que l'*échelle des répercussions défavorables* de l'ostréiculture en colonne d'eau et la *vulnérabilité du poisson et de l'habitat du poisson* correspondent à une activité ayant un risque faible qui a peu de probabilité de nuire de façon importante à la capacité de productivité ou à l'intégrité écologique. De plus, notre analyse suggère que les huîtres en aquaculture peuvent potentiellement jouer un rôle bénéfique dans ces systèmes et servir à combler plusieurs fonctions écologiques qui ont été perdues suivant les déclins historiques des populations d'huîtres. Étant donnée la nature de cette activité, nous concluons que les risques associés à l'ostréiculture dans la colonne d'eau peuvent être gérés de manière durable à l'intérieur d'un cadre de gestion adaptive qui comprend des mesures adéquates de planification et d'atténuation des impacts.

1 HABITAT MANAGEMENT QUALITATIVE RISK ASSESSMENT OF WATER COLUMN OYSTER AQUACULTURE IN NEW BRUNSWICK

1.1 Introduction

The Habitat Protection and Sustainable Development (HPSD) section of the Department of Fisheries and Oceans Canada (DFO) is responsible for evaluating potential environmental impacts on fish habitat associated with project development under the Habitat Management Program (HMP). DFO has been conducting environmental assessments of aquaculture impacts to fish habitat on a site-by-site basis under Section 35 of the *Fisheries Act* and coordinating the review of other federal authorities (FA) and expert authorities under the *Canadian Environmental Assessment Act* (CEAA). Given that the development of oyster aquaculture is among the growing activities in New Brunswick's (N.B.) coastal communities, the following qualitative risk assessment was conducted under the guidance of the HMP Risk Management Framework. This assessment of water column oyster aquaculture (i.e. suspended or off-bottom culture) integrates a thorough review of the relevant scientific information and a characterization of "works" (defined by CEAA) associated with oyster aquaculture, as it relates to fish and fish habitat *and the Policy for the Management of Fish Habitat*.

Risk is unavoidable and present in virtually every human situation. It is present in our daily lives, and in public and private sector organizations. The World Trade Organization (WTO) defines Risk Analysis as a "*systematic way of gathering, evaluating, recording and disseminating information leading to recommendations for a position or action in response to an identified risk*". Risk can be defined as a function of the probability of an adverse effect and the severity of that effect. In fact, this approach is used worldwide to manage the ever-changing uncertainties associated to human health, international trade, food safety, etc. (e.g. World Health Organization (WHO), WTO, Food and Agriculture Organization (FAO) Sanitary and Phytosanitary agreement, Hazard Analysis and Critical Control Point (HACCP)). Thus, a Risk Analysis is a tool intended to provide decision-makers with an objective, repeatable and documented assessment of the risks posed by an action. This approach recognises that every facet of life involves risks which can range from significant and adverse to negligible and inconsequential. Risks needs to be characterized, their significance assessed and thereafter managed to ensure a degree of comfort and control despite the uncertainties.

In context of the HMP Risk Management Framework, we define “Risk” as an event that has a specific likelihood of occurrence and identifiable impacts on the productive capacity of fish and fish habitat. A risk-based approach allows habitat managers to prioritize and focus efforts on regulating the activities which are considered to have the greatest potential impact to fish and fish habitat. This entails the review of available relevant information in order to categorize the risks associated with development proposals and associated management options. Through an objective and science-based decision-making process, activities are rated according to risk (e.g. low, medium and high) and then evaluated against the sensitivity of habitat and the scale of effects. This approach recognizes that high risk projects need to be managed differently than low risk projects. It is from this perspective that the following qualitative risk assessment of water column oyster aquaculture was prepared.

In collaboration with Maritimes Region and National Headquarters, a panel of scientists was brought together in 2006 under the *National Science Workshop: Assessing Habitat Risks Associated With Bivalve Aquaculture in the Marine Environment* National Assessment Process (NAP), to identify and characterize the potential environmental risks of bivalve aquaculture in the marine environment. The NAP was based on the peer review of working papers that addressed the identification, prediction, and measurement of the effects of marine bivalve aquaculture. The majority of the information presented at the workshop was based on the suspended culture of mussels on the east coast of Canada, but provided some indications as to the risk associated with bivalve culture in general. We have since undertaken the task of integrating the scientific advice which was relevant to water column oyster aquaculture into this Risk Assessment based on these frameworks and international definitions.

1.2 Regulatory context

In 1999, the Navigation Water Protection Program (NWPP) and CEAA recognized the need to consider aquaculture structures as having a fixed location and thus constituting a “work” under the *Navigable Water Protection Act* (NWPA). Therefore, these operations needed to be reviewed and approved under the NWPA. This led DFO to become a Federal Responsible Authority (FRA) for the review of aquaculture works under CEAA for the NWPP and a more formal federal review process which includes a fish habitat assessment under the habitat provisions of the *Fisheries Act*.

Following organizational changes in 2004, the responsibilities of FRA were transferred to Transport Canada (TC), with HPSD remaining involved on aquaculture files. To assist with that transition, DFO and TC developed a Memorandum of Understanding (MOU) whereby it was proposed that DFO help TC in the development of a Replacement Class Screening Report (RCSR) under section 19 of the CEAA to implement a more coherent approach in Environmental Assessment (EA) of these works. Rather than completing an EA for each project, the Act allows for the EA of some repetitive projects to be streamlined through the use of a class screening report. This signifies that if a project qualifies and meets the criteria set forth in the RCSR, it may not need an individual EA. This kind of report is built on and uses the knowledge accumulated through past environmental assessments of a given type of project. The class screening approach is considered compatible with an earlier proposal made by DFO to the New Brunswick Shellfish Aquaculture Environmental Coordination Committee (NBSAECC) operating under the 1995 Canada-New Brunswick MOU on aquaculture to develop an integrated shellfish aquaculture planning exercise. The Bay-by-Bay planning approach for aquaculture development was proposed to the Province of New Brunswick (aquaculture leasing and licensing is managed by the Province), in order to pre-define suitable areas for aquaculture based on an analysis of conservation and regulatory concerns of provincial and federal departments. It was presented to federal expert departments as a means to address cumulative impacts and inter-governmental regulatory concerns. The concept was accepted by the NBSAECC.

An initial pilot-project for the bays of Tabusintac and Richibucto was initiated in 2004. GIS databases were used to identify Valued Ecosystem Components (VECs) as well as potential conflicts with aquaculture works. Ecological reviews of the bays and layers of information, such as locations of bird colonies, avian species at risk, migrating and staging areas for waterfowl, fish habitat, wetlands, dunes, salt marshes, fisheries etc. were presented on maps. Potential use scenarios in conjunction with various management options were evaluated. This approach combined a number of GIS databases with current knowledge on user impacts to create an analytical tool to guide towards sustainable development. Zones were subsequently defined where shellfish leases could be best located to avoid potential spatio-temporal interactions with VECs. After a review of the pilot project results, the New Brunswick Department of Agriculture and Aquaculture (NBDAA) decided to continue the planning project, in collaboration with DFO and TC, for the remaining bays on the eastern coast of the Province.

1.3 Risk Analysis initiation

The current Risk Assessment expands on the scope of the evaluation of this activity and integrates the regulatory context which was required to support decision-makers in their review of water column oyster aquaculture works as they relate to fish and fish habitat. This is also compatible within the larger context of a Bay Management Framework developed in collaboration with the Province of New Brunswick. The geographic area for which the risk assessment was needed is Gulf New Brunswick (N.B.), but could also apply to Prince Edward Island (P.E.I.) and Gulf Nova Scotia (N.S.). In order to alleviate the remaining text, oyster aquaculture in N.B will refer to the Gulf portion along the eastern shore of N.B. and exclude the Bay of Fundy. The risk assessment was conducted to provide information to habitat managers about the potential effects of oyster aquaculture works and management options.

The format used for this assessment was inspired in part by the US Environmental Protection Agency Guidelines for Ecological Risk Assessment (ERA) (US EPA 1998). These types of tools are used to identify and characterize potential risks of the activity and to make a determination as to their significance as they relate to the productive capacity of fish habitat. In the HMP Risk Management Framework, this assessment is important for qualifying the residual negative risks after mitigation measures as well as subsequently determining options to manage the risks specific to the activity.

Additionally, because oysters in nature are recognized as providing beneficial ecological services and are often used as a compensation option for other works, a Net Environmental Benefits Analysis (NEBA) approach, as proposed by the US Department of Energy, was used to look at the potential gains minus the potential environmental costs of this activity (US Department of Energy 2003). Although the NEBA is not factored in to the HMP Risk Management Framework, we believe that a NEBA is consistent with the “Net Gain of Habitat for Canada’s Fisheries Resources” in the *Policy for the Management of Fish Habitat*. The policy states that the objective is to: “Increase the natural productive capacity of habitats for the nation's fisheries resources, to benefit present and future generations of Canadians”. We also believe that a NEBA can play a valuable role in considering the development of integrated management plans and in moving towards to DFO’s emphasis on an ecosystem approach.

The following diagram (Figure 1) illustrates how the two frameworks are used in parallel in this risk assessment on water column oyster aquaculture.

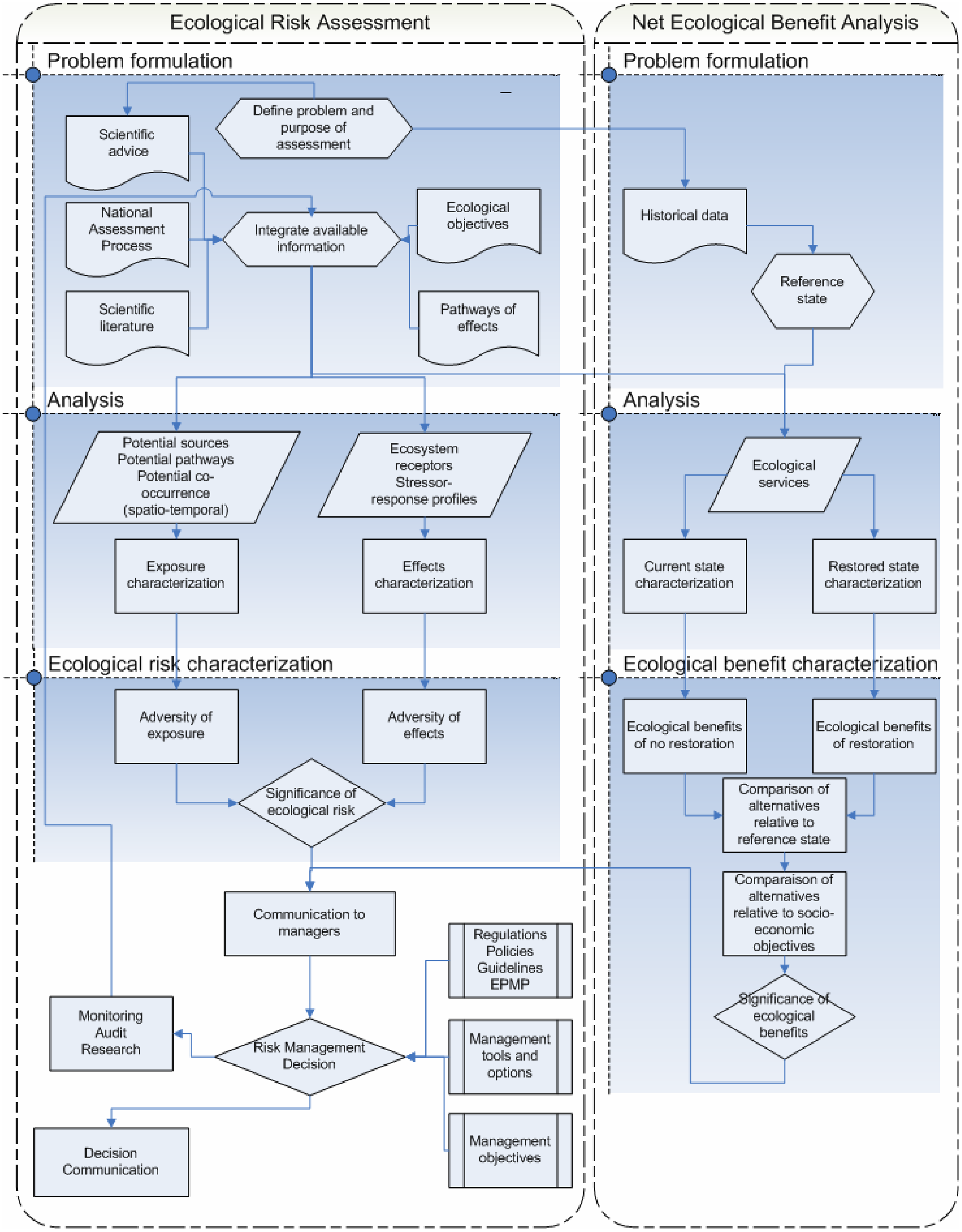

Figure 1 - Frameworks for Ecological Risk Assessment and Net Ecological Benefit Analysis

2 DESCRIPTION OF WATER COLUMN OYSTER AQUACULTURE

In the Maritimes, oyster culture is an activity which is usually practiced on a technically simple small-scale level. This activity is spread throughout coastal areas along the southern Gulf of St. Lawrence. In N.B, operations are mainly family owned; with a single proprietor for whom this is not their main occupation (75-90% of their income originates from other sources). The majority of these operations employ fewer than six employees, on a seasonal basis, but this number may range from one to sixteen employees. Most owners operate only one or two leases (Bastien-Daigle & Friolet 2006).

Procedures and activities associated with oyster culture in N.B. estuaries have a substantial history and record of development. Oyster aquaculture projects in New Brunswick have similar design, construction, operation and decommissioning characteristics. The following section summarizes the nature of the industry; the reader can consult Doiron (2006) for more detailed descriptions. Prince Edward Island and Nova Scotia use similar water column growing techniques. Prior to completing this risk assessment, a phone survey was conducted with individual growers to obtain an accurate picture of equipment and techniques currently in use (Bastien-Daigle & Friolet 2006).

2.1 Culture techniques

Unlike many parts of the world and the western region of Canada, where the exploitation of native species contributes little to commercial production (FAO 2005), the harvest and aquaculture of oysters along the Atlantic coast of Canada and the United States of America relies on a native species, the eastern oyster *Crassostrea virginica*. This species is found along the entire Northwest Atlantic seaboard, from Louisiana to N.B. with a large population in the southern Gulf of St. Lawrence (sGSL) (Kennedy *et al.* 1996).

In water column aquaculture, oysters are floated or suspended in the subtidal zones. Raising oysters above the substrate and placing them in bags or cages serves to enhance water circulation, water temperatures, and food availability. This in turn improves growth and decreases predation rates. Oysters grown in this manner are generally kept at low densities to help ensure that they can reach market-size within 3 to 4 years, rather than the 5 to 8 years normally required when grown on the substrate (DFO 2003b).

Presently, a variety of water column culture methods are used in N.B for growing oysters.; these include longline culture using bags, trays, or rope strings, or cages, and off-bottom culture using bags on French tables or on trestles. Provincial authorities define suspended culture as a form of aquaculture conducted in the water column or at the water's surface, where the structures are anchored but can float or move with the tides. They define off-bottom culture as being conducted in the water column where the structures are fixed in place on the substrate and do not move with the tides. The present risk assessment covers these two categories of techniques, commonly referred to as water column oyster culture. It does not include bottom culture, which is conducted directly on, or in, the substrate of an aquaculture site.

2.1.1 Suspended culture

Grow-out bags made of high density, UV-resistant polymer mesh (often referred to by the manufacturer's name, such as Vexar™ or Durethene® bags) are used to contain the oysters. The bags are either equipped with individual floats and attached to a longline system or inserted in a cage structure equipped with floats. Bags measure 85 cm (long) by 40 cm (wide) by 10 cm (high). The density of oysters in the bags is progressively reduced over the 3-4 year grow-out period as the oysters grow (Doiron 2006). Initially, 15-25 mm oysters are placed at densities of 1000-1500 oysters per bag (2-3 kg). In the final year of production, oysters typically measure 50-75 mm and are held at densities of 200-250 per bag (4-6 kg) to ensure adequate growth and a desirable shape (i.e. choice or fancy grade rather than commercial or standard) (Doiron 2006).

In the longline system, grow-out bags are lie flat on the surface of the water with one buoy on each side and secured by parallel lines anchored to the bottom (Figure 2). The most common design usually consists of two rows of approximately 50 floating bags, but many variations of this system can be observed. Two main anchors maintain the longline in a fixed location; these consist of concrete blocks, metal anchors or screw anchors. The lines are kept separated by spreader bars installed approximately every ten bags. Growers can adjust the buoyancy of the grow-out bags by changing the location of the buoys on the bags. Each longline system measures approximately 60 m from anchor to anchor, and is spaced 6-10 m from other longlines to provide water circulation around the bags and boat access for regular maintenance. Growers typically install 15 to 20 longlines per hectare. Longlines are usually oriented along the most appropriate axis to reduce wear from tides and currents on equipment.

Cages are made of a plastic coated wire-meshed material (similar to the Aquamesh used in many lobster traps) and are designed to contain between 2 to 6 grow-out bags; six being the most common configuration. The grow-out bags are placed in divided sections of the cage, which function as drawers. In order to ensure adequate water circulation, no more than two bags are placed over one another. Each cage is equipped on the upper side with two buoys allowing it to float immediately below the water surface. Buoys can be made of a variety of materials, including Styrofoam and PVC. The cages are secured either by using single anchors or by attaching them to longlines. Generally, growers will install 12 cages per 50 m longline with a maximum of 20 lines per hectare (240 cages/ha). As above, lines are separated by a corridor to allow boat access.

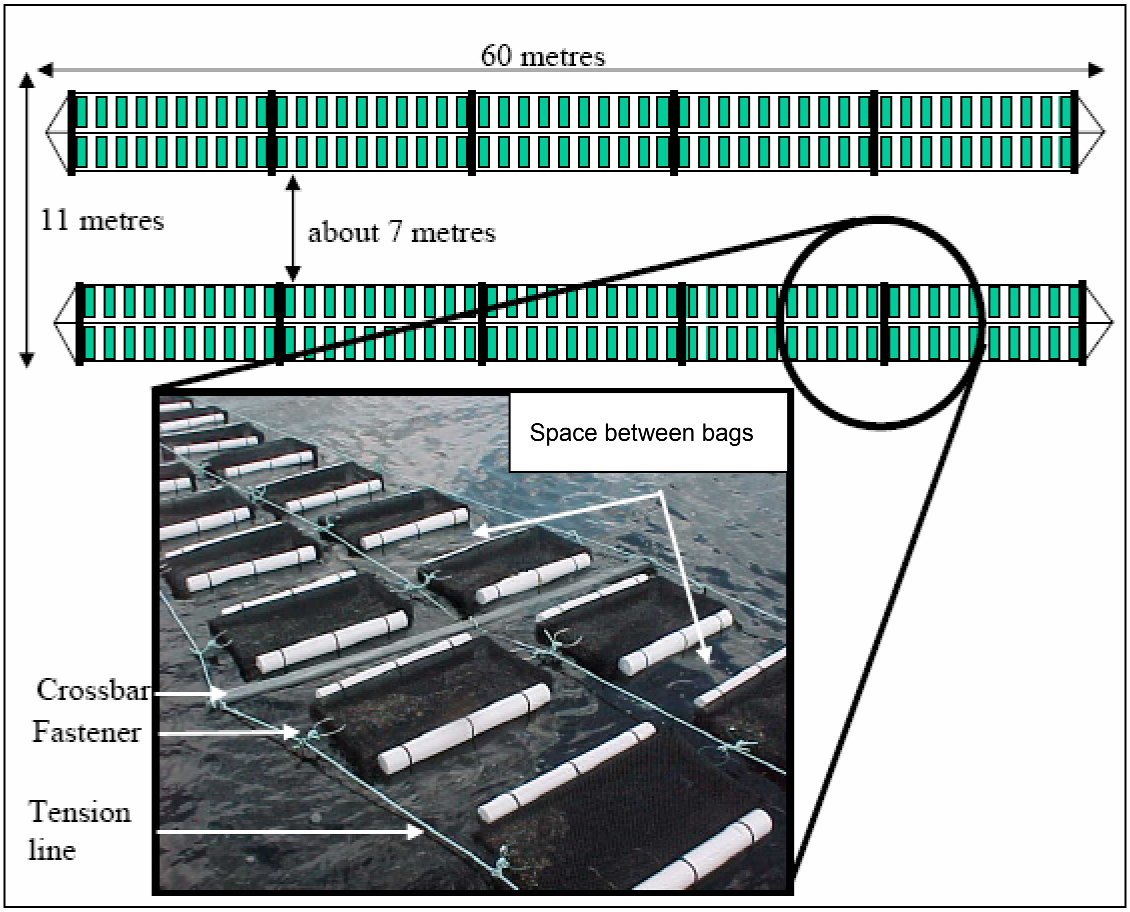

Figure 2 - Description of the longline structures used in N.B. (modified from Doiron 2006)

A less common suspended technique is known as rope culture, whereby clusters of oysters are attached directly to a rope at regular intervals (without any housing). Ropes are suspended in the water column or floated at the water surface level using specifically designed supports, which function similar to longline systems. Oysters cultured on rope remain submerged at all times.

2.1.2 Off-bottom culture

Oyster tables, also known as French tables, consist of a metal rod structure on which grow-out bags containing the oysters are supported. This platform raises the bags sufficiently to ensure water circulation around the oysters. Depending on the site, oysters can be uncovered during each tidal cycle or remain constantly submerged. Another off-bottom technique consists of raising the oyster bags on runners or pipes placed on the sediment. Both of these techniques typically require setting the structures in sections of the lease with little or no eelgrass to ensure proper water circulation. Oyster tables and runners are removed at the end of the growing season to avoid ice damage.

2.1.3 Site preparation

Unlike some types of on-bottom shellfish aquaculture that require extensive bottom preparation (e.g. dragging, additions of gravel, dredging, removal of vegetation, etc.), no specific site preparation activities are required for water column oyster aquaculture sites other than installing the equipment and anchors.

2.2 Installation

The installation of structures is generally done from a boat or from the ice surface during winter. For longlines, anchors are installed either directly on the marine sediment (concrete blocks) or driven into the sediments (anchors). In general, the anchoring system is designed to be permanent. French tables and runners are installed directly on the substrate but are removed seasonally.

2.3 Operation

Maintenance of the inventory includes stock rotation and reducing the density of oysters to ensure optimum growth and quality; this may occur 2 to 3 times during the growing season. As the bags float at the surface of the water, with one side submerged and the other exposed to air, fouling by epifaunal plants and animals can be removed simply by turning the bag (180°) to allow the attached organisms to desiccate or by pressure washing. The frequency of this maintenance depends on the growth of epifauna which varies during the season; being more pronounced in the summer and less so in the fall. In general, air drying takes a few days. Oyster culture does not require food supplements, treatment with pharmaceuticals, disinfectants, or hormones.

2.4 Oyster spat collection

Oyster spat or oyster seed can be collected by the producer or can be purchased from other local growers. Oyster seed can only be collected from approved oyster collection areas or on private leases. Oysters typically spawn between early and mid-July depending on latitude and annual conditions. A variety of collectors are used to attract oyster larvae, which preferentially settle on clean and textured surfaces. It is critical to deploy these collectors in the appropriate areas at the correct time. After approximately two to three weeks of drifting in the currents, competent larvae cement themselves to the collector's surface. Afterwards, when oyster seed reach a sufficient size, the collectors are transferred to the lease (if seed are not collected from the lease area itself). Depending on their size, the seed oysters are stripped from the collectors in the fall or the following spring, sorted by size and transferred to the grow-out bags.

2.5 Overwintering

In much of gulf N.B., the upper water column freezes in winter. In order to protect the oysters, structures must be overwintered in below the depth to which the ice can extend or in areas that are not prone to ice jams, or frequent ice movement. Typically, oysters are moved to the deepest portion of the aquaculture site and sunk to the bottom during the winter months. This period corresponds to a period of dormancy for the oysters, where filtration and feeding effectively stop.

Oysters are overwintered in bags or cages. The longlines can be either submerged below the surface, deep enough to avoid the ice, but not touch the seabed (using weights to counter the buoyancy of the equipment), or the floats are removed from the bags/cages and the structures are allowed to lie on the substrate. Sunken lines are located by GPS or by triangulation to facilitate retrieval during winter harvesting or for re-suspension. Oysters are re-deployed to the grow-out site the following spring; re-suspension is carried out as soon as possible after ice break-up.

2.6 Harvesting

Harvesting occurs when oysters reach marketable size. During the ice-free period, harvesting is generally done by boat; grow-out bags are light enough to be removed by hand from the structures and loaded onto vessels. The heavier cages may require a winch to hoist them onto the boat. The transport boat typically unloads the bags and product at a landing from where it is delivered by truck to a processing facility.

During the winter harvesting, the overwintering sites are typically accessed by all terrain vehicles or snowmobiles. An access hole is cut through the ice with a chain saw or auger and a portion of the stock is retrieved manually or with the use of manually-operated hydraulic equipment. Divers may be required to assist in retrieval of the stock.

2.7 Predator Control

Predators are of greatest concern during the spat collection phase when oysters are small and not protected within the grow-out bags. In some cases, predators such as crabs and starfish are controlled by dipping the collectors for a few seconds in a freshwater or diluted lime bath. Competitors or predators found within the grow-out bags are manually removed during regular maintenance activities.

In gulf N.B. oyster culture, there are no control measures which could harm marine life such as birds or mammals (i.e. anti-predator nets, acoustic scaring devices, etc.). The need for predator removal is rare in the case of off-bottom oyster culture, because the stock is protected within the grow-out structure.

2.8 Decommissioning

Within 90 days of cessation of aquaculture activities, the holder of the aquaculture occupation permit or the aquaculture lease is required under provincial jurisdiction (N.B. *Aquaculture Act*, 1988, c. A-9.2, and 91 158 of the N.B. Regulation under the *Aquaculture Act*) to restore the site to the satisfaction of the Minister. If the holder does not restore the aquaculture site within the prescribed time or in a manner considered satisfactory by that authority, NBDAA will have the site restored, and the holder will be liable for all restoration costs.

3 RISK MANAGEMENT FRAMEWORK

The Risk Management Framework is intended to provide a structured approach to decision-making that takes into account the concepts of risk, uncertainty and precaution. A Risk Assessment is a process used to determine the level of risk that residual effects pose to fish and fish habitat based on the information currently available. Risk Assessments are used to determine the technical parameters that are useful and feasible for risk management.

To assess risk to fish and fish habitat, one must consider the severity of the effects in the context of the sensitivity of fish and fish habitat being affected by the activity. The Risk Assessment Matrix (Figure 3) incorporates these two factors in order to characterize the level of risk posed by the development proposal on the productive capacity of fish habitat. The rationale used to locate the residual effects on the matrix forms the basis for decision-making.

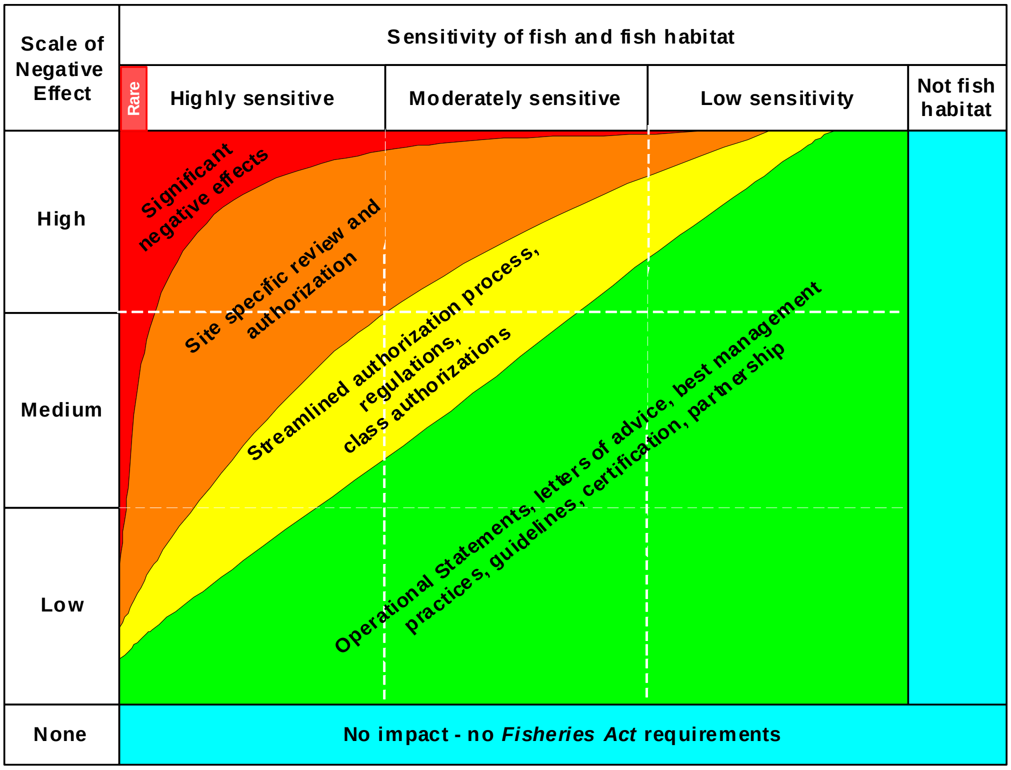

Figure 3 - DFO Habitat Management Program's Risk Assessment Matrix

3.1 Scale of negative effects

The following attributes are used to scale residual effects on the y-axis of the risk assessment matrix (Table 1) and are adapted to aquaculture. Ratings are assigned to evaluate the predicted effect of the activity. For every effect, the degree of adversity of each attribute is assessed and this helps to determine the overall residual effect significance.

Table 1 - Attributes used to describe the scale of negative effects to fish habitat

Criteria	Importance level rating		
	Low	Medium	High
Magnitude	Localized effect on specific group, habitat, or ecosystem, returns to pre-Project levels in one generation or less, within natural variation	Portion of a population or habitat, or ecosystem, returns to pre-Project levels in one generation or less, rapid and unpredictable change, temporarily outside range of natural variability	Affecting a whole stock, populations, habitat or ecosystem, outside the range of natural variation, such that communities do not return to pre-Project levels for multiple generations
Geographic Extent	Limited to aquaculture footprint and vicinity	Limited to aquaculture lease and vicinity	Extends beyond the aquaculture lease area
Duration of Effect	Less than one season	Less than one year	A year or longer
Frequency of Effects	Occurs on a monthly basis or less frequently	Occurs on a weekly basis	Occurs on a daily basis or more frequently
Reversibility	Effects are reversible over short term without active management	Effects are reversible over short term with active management	Effects are reversible over extended term with active management or effects are irreversible

3.2 Sensitivity of fish and fish habitat.

The sensitivity of fish and fish habitat (represented by the x-axis of the matrix) can be defined in relation to the degree and duration of damage caused by a specified external factor. Sensitivity may refer to the structural fragility of the entire habitat in relation to a physical impact, or to the intolerance of individual species comprising the habitat to environmental factors, such as exposure, salinity fluctuations or temperature variation.

Habitat can be defined as "the structural component of the environment that attracts organisms and serves as a center of biological activity" (Peters & Cross 1992 cited in Auster & Langton 1998). In this example, habitat would include the range of sediment types (e.g. mud, sand, pebble, etc.); and bed forms (e.g. sand waves and ripples, mudflats, etc.) as well as the co-occurring biological structures (e.g. shell, burrows, submerged aquatic vegetation, etc.). Defining sensitivity for all these components is problematic. Ideally, models of sensitivity indices for specific habitats, communities, and key taxa-based on the effects of specific activities, levels of effort, and life history patterns (of both fish and taxa which serve a habitat function) would be developed (Auster & Langton 1998). Such indices are not currently available; as a substitute, the *Habitat Management Policy* recommends the use of a matrix analysis to determine the sensitivity of fish and fish habitat.

This matrix uses general qualifiers to describe fish and fish habitat attributes (summarized in Table 2). Sensitivity is defined in terms of species or habitat susceptibility to changes and perturbations as result of an activity or modifications in environmental conditions, such as suspended sediments, water temperature or salinity. Dependence is defined in terms of the use of habitat by fish species; for example, some species may be able to spawn in a wide range of habitats, while others may have very specific habitat requirements. Rarity is defined in terms of the relative strength (abundance within a range) of a fish population or the prevalence (ecological redundancy) of a particular type of habitat in a community. Resilience refers to the ability of an aquatic ecosystem to recover from changes in environmental conditions.

Table 2 - Attributes used to define sensitivity of Fish Habitat

Criteria	Importance level rating		
	Low sensitivity	Moderately sensitivity	Highly sensitivity
Sensitivity	Species/habitat present are not sensitive to change and perturbation	Species/habitat present are moderately sensitive to change and perturbation	Species/habitat present are highly sensitive to change and perturbation
Dependence	Not used as habitat; or used as migratory habitat only	Used as feeding, rearing, and/or spawning habitat	Habitat critical to survival of species
Rarity	Habitat/species is abundant within its range or community; ecological redundancy is widely present	Habitat/species has limited distribution; is confined to small areas; ecological redundancy is present	Habitat/species is rare; ecological redundancy is absent
Resiliency	Species/habitat is stable and resilient to change and perturbation	Species/habitat is stable and can sustain moderate level of change and perturbation	Species/habitat is unstable and not resilient to change and perturbation

4 ECOLOGICAL RISK ASSESSMENT

Ecological risk assessment is based on the characterization of the potential effects and characterization of exposure (US EPA 1998). Effects are linked to ecosystem receptors and stressor-response profiles. Exposure is linked to potential pathways of effects, potential sources and potential co-occurrence. Exposure is also related to the scale and intensity of activities. The scope of this ecological risk assessment focuses on water column oyster aquaculture as it relates to fish and fish habitat.

4.1 Effects characterization

4.1.1 Potential pathways of effects

The analysis of the potential pathways of effects is largely based on information contained in the NAP documents (Anderson *et al.* 2006, Chamberlain *et al.* 2006, Cranford *et al.* 2006, DFO 2006, McKindsey *et al.* 2006a, Vandermeulen *et al.* 2006) as well as the Statement of Knowledge (SoK) reports (DFO 2003a). These various papers, which undertook comprehensive reviews of the science available, provide extensive details on shellfish aquaculture in general and aid in the specific characterization of the potential effects of water column oyster aquaculture. Consequently, the following sections only discuss some of the major points in a cursory manner.

Many of the adverse effects and concerns in the conclusions from the NAP were linked to studies conducted in Tracadie Bay, P.E.I. Much of the discussion and most of the modeling results presented focused on the evaluation of carrying capacity for this bay, which is one of the most intensively cultured and studied bays for shellfish aquaculture in the Gulf Region. Approximately 40% of Tracadie Bay’s surface is leased for mussel cultivation, with an annual mussel production of 2,000 t. From 1990 to 2001, the leased area grew from approximately 20% to 40%, while the biomass of mussels increased by over 300%. This corresponds to an atypical scenario and is not considered entirely representative of other bays or other types of shellfish production in the region. Tracadie Bay has thus become a focal point for research on the negative environmental effects of shellfish aquaculture. However, it remains unclear as to the net effects of the culture on the overall productivity of the bay even in these circumstances. Miron *et al.* (2005) found that the absence of a strong relationship between husbandry practices and the studied benthic parameters might be related to the oceanographic characteristics and land-based activities associated with the water system rather than direct and cumulative effects of mussel culture. Nonetheless, the NAP highlighted a series of concerns with regards to bivalve aquaculture in general which are useful in this analysis. The reader may also refer to the documents listed above for more information on benthic and water column effects.

Potential effects can be linked to the presence of oysters and the presence of structures in the water. In the particular case of oyster aquaculture, one must also understand the functional effects of natural oyster populations in an attempt to understand their role in aquaculture operations. Interactions in the coastal zone between farmed bivalves and other organisms are highly complex. Net habitat effects of bivalve aquaculture are difficult to disentangle from effects of other anthropogenic activities (McKindsey *et al.* 2006a). In addition, net pathways of effects on the environment can be both negative and positive. Figures 4 and 5 represent simplified views of some of the complex ecological interactions that can occur in relation to bivalve aquaculture. The scientific literature indicates a variety of levels of effects of bivalve farming activities on the many compartments of estuarine ecosystems.

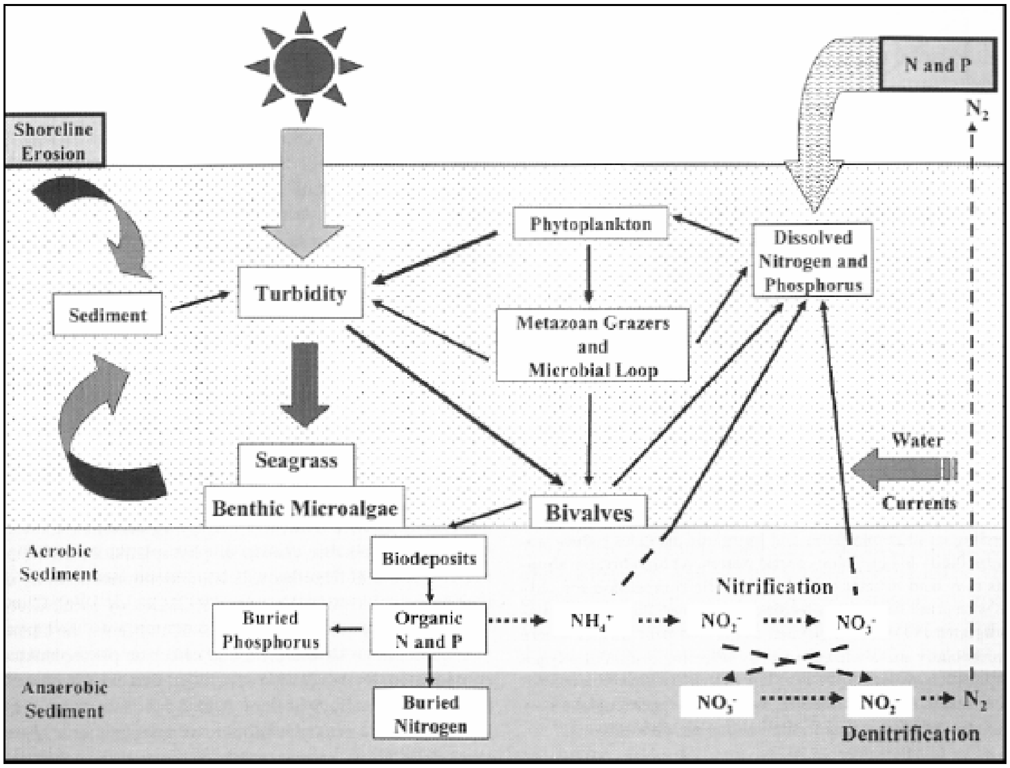

Figure 4 - Conceptual diagram of the ecosystem effects of suspension-feeding bivalves. Solid lines indicate transfer of materials; dashed lines indicate diffusion of materials; dotted lines indicate microbially mediated reactions (Vandermeulen 2006 from Newell 2004).

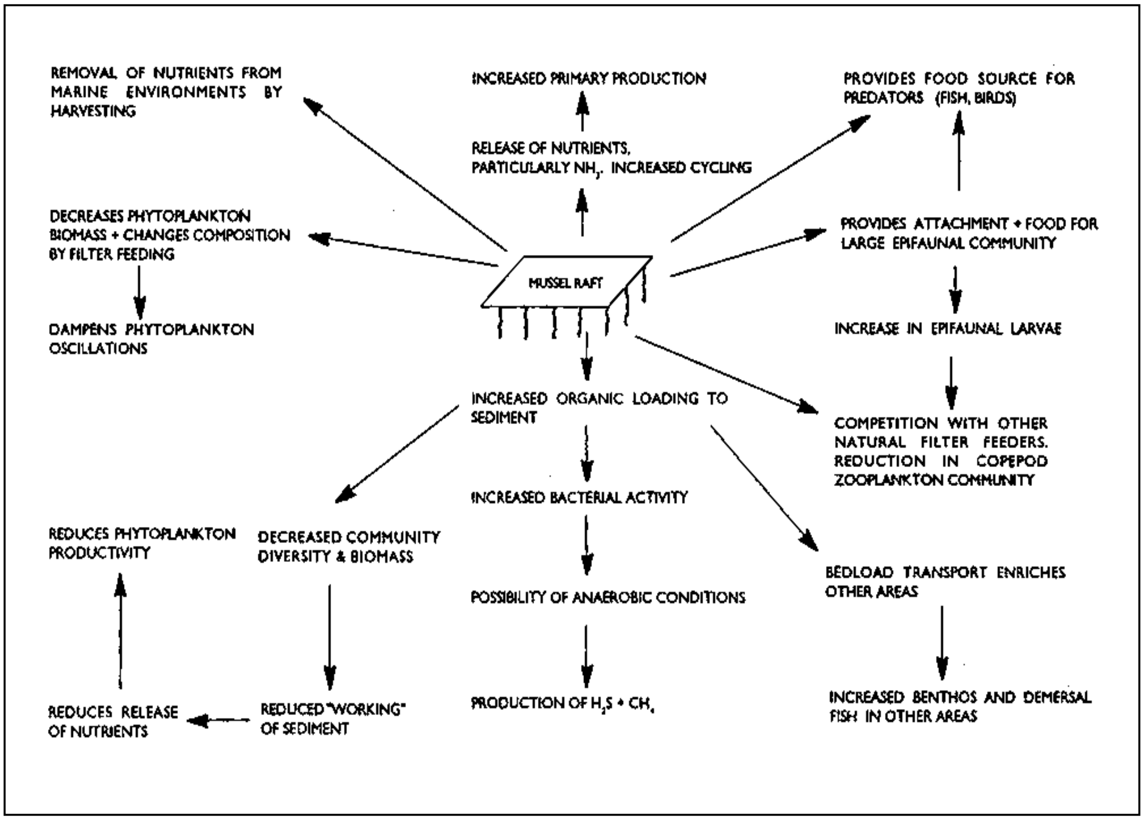

Figure 5 - Potential pathways of effects from mussel culture systems; (FAO 2006)

4.1.2 Potential sources

Potential sources of effects that can be expected from shellfish aquaculture have been identified by ICES (2004) and only the effects relevant to fish and fish habitat are summarized in the table below (Table 3).

4.2 Exposure characterization

4.2.1 Adversity of exposure and effects

DFO used the lists of pathways of effects and endpoints of concerns to scope potential interactions between oyster aquaculture and Valued Ecosystem Components (VECs). A compilation of mitigation measures currently requested of the industry to protect fish habitat was done and applied in the analysis of effects (Table 4). The information provided by the NAP, scientific literature, and monitoring results was also used in the evaluation of the potential residual negative effects to fish habitat.

Table 3 - Summary of steps in bivalve aquaculture and their potential to influence fish habitat. Based on ICES 2004 and adapted to N.B. water column oyster aquaculture.

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1. Seed collection

- a. artificial collectors
  - i. removal of juveniles from wild population of target species
  - ii. increasing recruitment success of oysters or other species
  - iii. alteration of the hydrodynamic regime
  - iv. acting as fish attraction device (FAD)

2. Grow-out

- a. effects common to all techniques
  - i. organic enrichment of seafloor
  - ii. alteration of hydrodynamic regime (current speed, turbulence)
  - iii. food web effects: competition with other filter feeders, increasing recycling speed of nutrients
  - iv. providing food for predators of shellfish
  - v. control of predators and pests
  - vi. acting as artificial reef or FAD (attraction/displacement or enhancement of animals)
- b. artificial structures (trestles, poles, rafts, longlines)
  - i. risk of attraction of birds
  - ii. risk of damage to eelgrass

3. Harvesting

- a. effects common to all techniques
  - i. removal of biomass/nutrients
  - ii. removal of non-target species
  - iii. competition with predators
- b. collection of off-bottom structures
  - i. risk of trampling of substrate and vegetation

4. Processing

- a. effects common to all techniques
  - i. discard of epibionts
  - ii. discard of shells

Table 4 - Review of potential ecological concerns of water column oyster aquaculture.

PATHWAY OF EFFECT	***CONCERN***	***POTENTIAL EFFECT***	***DURATION OF EFFECT***	***MITIGATION / OBSERVATIONS***
Addition of physical structure in the water column	Changes in physical environment	Physical structures can modify the hydrodynamic patterns of water movements by impeding or altering water flow.	Grow-out period	• Site infrastructure is required to be aligned with dominant currents to minimize impacts on water flows. • Minimal spacing recommended between structures of 3 m (industry currently spaces structure 7-10 m apart) • NWPA prohibits works in navigation channels. • Structure is considered permeable to fish and marine mammals, no leader, net or entrapment mechanism that could impede migration or organism movement. • No leader, lures, nets or other obstacles that could impede movement, cause entanglement or attract predators.
		Physical structures can change flow patterns and increase sedimentation under the structures.		
		Physical structures may become obstacles for the movement or reproduction of organisms.		
		Overwintering of physical structures may affect benthic fauna or flora.	Overwintering period	• Minimal concern as bags is typically overwintered during the period of dormancy for most organisms. • Overwintering is generally conducted in deeper waters where presence of flora is limited. • Re-suspension is done as early as possible after ice-out to reduce losses.
	Changes affecting species composition	Physical structures in the water column may displace certain organisms from the footprint of the structure.	Grow-out period depends on local husbandry methods and faunal community	• In water column oyster aquaculture, the footprint which can exclude organisms from an area is considered minor. • Oysters not available to predation within grow-out bags. • No lures or bait that could attract predators or scavengers. Oysters not within diet of large marine predators, such as seals. • Presence of epibionts on or falling off structures may attract crustaceans, fish and birds.
		Physical structures in the water column create habitat for organisms by providing a substrate similar to the effect of an artificial reef.		

PATHWAY OF EFFECT	*CONCERN*	*POTENTIAL EFFECT*	*DURATION OF EFFECT*	*MITIGATION / OBSERVATIONS*
		Structures provide a hard substrate for opportunistic organisms or also for colonizing organisms which can serve as food for fish and invertebrates.		• Preliminary studies suggest that species diversity near structures appears to be maintained although the species composition may be altered. • Proponent has to select its site, deploy its structure and adopt appropriate husbandry practices to minimize colonization of other organisms.
		Structures may affect aquatic *Species At Risk Act* (SARA).	Grow-out period	• No species currently listed in N.B. estuaries. • Potential risk of spatio-temporal interaction between water column oyster aquaculture and aquatic SAR is not significant given the spatial area where culture occurs.
	Changes in light penetration	Physical structures in the water column may reduce the light availability to flora (i.e. eelgrass) directly under the structures.	Tidal dependant	• Siting of off-bottom aquaculture in eelgrass-free areas. • Minimal spacing of off-bottom aquaculture works at minimum of 3 m, not to exceed 50% coverage of the site. (industry currently spaces structure 7-10 m apart) • Suspended aquaculture is to be anchored to allow swaying with each tidal cycle and to avoid continuous shading of the same area of eelgrass. • Structures are to be designed and installed to maximize opening to increase light penetration. • The footprint of structures on the benthos is small.
Addition of filter feeding bivalves	Changes in population interactions	The oysters maintained in water column aquaculture may reproduce with wild populations of oysters.	Spawning period	• Not a concern given that the oysters are recruited yearly from wild sources and not from hatcheries.
		The addition of oysters may cause a competition for space with other organisms.	Grow-out period	Not expected to be an issue given that oysters are held in the artificial structures in the water column which create additional space.

PATHWAY OF EFFECT	*CONCERN*	*POTENTIAL EFFECT*	*DURATION OF EFFECT*	*MITIGATION / OBSERVATIONS*
		The addition of filter feeding bivalves in the water column may cause removal of eggs and larvae of fish and benthic organisms.	Sporadic, during egg and larval stages – within size preference for oysters	• Not expected to be an issue given that *C. virginica* is the native species of oyster and population interactions with that species are expected to be similar in aquaculture as they are under natural conditions. • Demonstrated preference by oysters for microzooplankton as opposed to mesozooplankton. • Narrow range of opportunity if within immediate vicinity of feeding current vs. total surface of estuaries. • Adaptation mechanisms within bivalve populations to limit egg and larval predation of con-specifics. • Observed presence of higher diversity of species within natural oyster beds (including other bivalves).
		The addition of oysters to the water column may attract predators.	Seasonal	• Oysters are protected within the grow-out bags, except for a limited time while on collectors. • Fouling organism fall-off from growing structure may add food to benthos. • Additional gametes and larvae may contribute to food web. • No documented evidence of large predators near these sites. • Not a preferred food-source for large predators.
	Changes in plankton abundance	The additional biomass of filter feeding bivalves to the water column may cause a depletion of plankton.	Grow-out period	• Not expected to be an issue given that *C. virginica* is the native species of oyster and population interactions with that species are expected to be similar in aquaculture as they are under natural conditions. • Current densities are lower than historical densities found in natural populations throughout the region.

PATHWAY OF EFFECT	***CONCERN***	***POTENTIAL EFFECT***	***DURATION OF EFFECT***	***MITIGATION / OBSERVATIONS***
	Changes in water quality	The addition of filter feeding bivalves to the water column may remove significant quantities of particles from the water column that can reduce turbidity.	Grow-out period	• This effect is largely considered to be beneficial by reducing turbidity, thus favouring the growth of aquatic vegetation.
	Changes in nutrient cycles	The addition of filter feeding bivalves to the water column may play a significant role in recycling nutrient and benthic/pelagic coupling.	Grow-out period	• This effect is largely considered to be beneficial by removing excess nutrients through bivalve feeding as well as harvesting.
	Changes in organic enrichment	Biodeposition from faeces and pseudofaeces may increase sedimentation and enrich the benthos which could affect benthic geochemistry and organisms.	Grow-out period	• Not expected to be an issue under current stocking oyster densities and given seasonal nature of operations. Bays where water column oyster aquaculture sites occur in N.B. are characterised as dynamic shallow water systems with frequent resuspension of upper layers of sediment by wind, wave, tides, storm-events and ice-scour which likely reduce the effect of biodeposition.
Husbandry Activities	Changes caused by equipment installation	Equipment installation and regular maintenance activities at the site may temporarily increase turbidity	Sporadic, during installation and maintenance activities	• Access to the intertidal zone by motor vehicles other than boats is prohibited under provincial regulations, unless operating such vehicle on ice or frozen ground that is completely covered by snow.
		May cause physical damage to the eelgrass.		• Anchors are to be sized and installed to minimize dragging, preferably during winter (eelgrass dormant period). • Trampling, anchoring in eelgrass, are to be minimized.
	Discard of epibionts	Discards of epibionts during maintenance may be deposited to the benthos.	Sporadic during maintenance activities	• Air-drying of the equipment through bag turning is the recommended method of removal in the aquatic environment. • Disposal and recycling of waste on land is controlled by provincial and municipal regulations

PATHWAY OF EFFECT	*CONCERN*	*POTENTIAL EFFECT*	*DURATION OF EFFECT*	*MITIGATION / OBSERVATIONS*
	Discard of shells	Discards of shells during maintenance may be deposited to the benthos.	Sporadic during maintenance activities	• Not expected to be a significant issue, discouraged by industry to prevent spread of boring sponge (*Cliona celata*) • Incidental loss of small quantities is considered positive for habitat creation
	Use of artificial food, pharmaceuticals or chemicals	Potential to release undesirable compound into the environment during production or cleaning activities.	Grow-out period	• Bivalve aquaculture does not require the use of artificial food, pharmaceuticals or chemicals for production purposes. • Air drying is the typical method for cleaning equipment in the aquatic environment. Pressure washing with water is also used although less frequently. These methods do not require chemical cleaning agents. • Use of lime bath to remove predators on collectors is sporadic.

4.2.2 Scale and intensity of exposure

Concerns with regards to the adverse effects of bivalve aquaculture appear to be linked to the scale and intensity of aquaculture rather than the type of culture or infrastructure used (McKindsey *et al.* 2006b). In aquaculture, the scale and intensity is typically related to the rearing density of the animals (numbers per area) and to the extent of the activity (area occupied) (i.e. the level of exposure). Exposure is a function of sources, distribution and co-occurrence in space and time between an effect and the receiving environment. The following sections attempt to characterize the scale and intensity of oyster aquaculture in the sGSL.

4.2.2.1 Oyster production in the Maritimes

It is difficult to obtain precise landing values from oyster aquaculture in the Maritimes because of the way statistics on oyster production are collected and estimated. For instance, DFO keeps records of oyster purchases, as reported on sales slips, including data on both commercial wild-harvested and aquacultured oyster statistics and it is not currently possible to disentangle the respective proportion of cultured versus fished oyster from the values reported.

The Province of N.B. estimates aquaculture production based on an assessment of the number of oyster growing bags in use. In 2004, for example, the Province estimated that 165,000 oyster bags were in production, with an average of 500 oysters per bag, which would have signified approximately 82.5 million oysters (Government of New Brunswick 2004). Only one fourth of these would have been available for harvest (production time of 4 years), which would amount to 20.6 million harvestable oysters (approximate size of 60 mm @ 39.10g/oyster for an approximate total of 805t) (Government of New Brunswick 2004). Robichaud (unpublished) conducted an audit of oyster aquaculture leases in N.B. in 2006 and arrived at a slightly lower estimate of approximately 140,000 bags.

A comprehensive survey (interviews, boat and aerial photography) of oysters under production in N.B. concluded that between 990 and 1,249 tonnes of oysters (all sizes included) were under cultivation in 2005 (Comeau *et al.* 2006). The discrepancy in production estimates between the three main sources of information (producers, government officials and sales slips) illustrates some of the difficulty in quantifying actual production. Comeau *et al.* (2006) estimated the actual production of marketable oysters in 2005 to have been 679 t from aquaculture and 75 t from commercial harvesting, which puts the estimated total landings at 754 t.

4.2.2.2 Scale of oyster aquaculture production

According to Morse (1971) interest in oyster farming, characterized by an expansion in the number of leases and the development of seed production facilities and assistance programs, began in earnest in the Maritimes in the 1940's. Twenty years later, in 1966, it was estimated that 87% of the total landings of oysters could be attributed to the public fishery and 13% from public lease production.

Attempts have been made to project future landings by Unic Marketing which estimated that the future contribution of aquaculture would gradually begin to increase and that it would equal the contribution from the wild fishery by 2010 (Unic Marketing Group Ltd 2003). However, based on the numbers above, it appears that these predictions have failed to materialize and that aquaculture production remains below expectations. Landings from aquaculture production may only be gradually replacing commercial landings, perhaps because natural oyster reefs continue to be depleted (C. Noris, personal communication) and/or the industry may not be expanding as rapidly as initially predicted.

4.2.3 Relative intensity of aquaculture production

The intensity of aquaculture production has been equated with densities of bivalves under production for a specific surface area, or annual yield. Moreover, the culture intensity and yields speak in part to the concept of carrying capacity.

Comeau *et al.* (2006) calculated that average densities of oysters grown in N.B. were seven times lower than densities used in Normandy, France. The biomass of oysters (0.23 kg/m^2 of leased area) in N.B. by comparison to mussels or with oysters cultured in other areas in the world (10 – 85 kg/m^2) is considered to represent a low intensity production (Comeau *et al.* 2006). In Spain's Rias Bajas, one raft (average 19 x 16m) is estimated to produce 50 metric tons, or 164 kg/m^2 (Tenore *et al.* 1982). This is one of the highest reported protein yields per unit area and is only possible given the nutrient-rich upwelling conditions and high primary productivity observed in this region. To illustrate the range of densities used in oyster aquaculture, the following table (Table 5) shows oyster densities reported in the literature, along with reported environmental effects. By comparison to the scale and intensity of these operations, oyster aquaculture densities used in the Maritime Provinces, which are among the lowest described in the literature, constitute a low-intensity culture situation.

The comparison of yields and reported effects also provides some indications of thresholds of exploitation, as well as site-specific environmental conditions, that can occur and in which detectable and significant negative impacts can be observed. We are unaware of any study which can demonstrate significant adverse effects of bivalve culture at the densities observed in New Brunswick water column oyster aquaculture.

It is also interesting to note that the transition to off-bottom culture resulted in an actual reduction of stocking densities of oysters compared to on-bottom operations and natural oyster reefs. Moreover, oyster densities in natural reefs are estimated to have been 17 to 530 times greater than those currently measured in aquaculture (Comeau *et al.* 2006). Oysters in natural and healthy oyster reefs (Table 6) occur at densities in excess of hundreds of oysters/m^2 (500 – 4,000 oysters/m^{2,} roughly equivalent to 25 to 55 kg/m^2) (DeAlteris 1988; Paynter 2002; Harris 2003).

Table 5 - Yields of oysters produced in aquaculture, from temperate ecosystems, converted when applicable to a standard equivalent of metric tonnes per hectare per year (McKinnon et al. 2003)

Species	Region, country	Average yield t ha^{-1} yr^{-1}	Method	Reported environmental effects			Reference
				Benthic infauna / epifauna	Organic / inorganic loading	Redox / sulphides	
C. gigas	Tasmania, (Australia)	20	Longlines	No significant differences in benthic infauna	No significant trends in organic carbon along farm transects	No negative redox measurements found beneath farms	(Crawford *et al.* 2003)
C. gigas	River Exe, (England)		Trestles	Decreased abundance of macrofauna (half) restricted to footprint	Increased sedimentation rate, increased organic content (footprint)	Reduction in depth of oxygenated layer (footprint)	(Nugues *et al.* 1996)
C. gigas	Arcachon (France)	13	Tables	Increase in meiofauna abundance (3-4 times) and decreased macrofaunal abundance (half)	Elevated organic carbon levels (footprint)	Elevated oxygen demand and anoxic conditions	(Castel *et al.* 1989)
C. gigas	Thau (France)	10	Rafts, "semi-intensive"				(Chapelle *et al.* 2000)
C. gigas	New Zeland	8	Racks	No marked trend in macrofauna species richness, species composition and dominance patterns	More elevated sedimentation directly under racks	No evidence of highly enriched conditions	(Forrest & Creese 2006)
C. gigas	B. C. (Canada)	4					http://www.agf.gov.bc.ca/fish_stats/statistics-aqua.htm
C. virginica	NB (Canada)	4	Tables	Macrofauna biomass, abundance and number of species higher or similar	No organic enrichment	Seasonal variations but no significant differences between control and culture sites	(Mallet *et al.* 2006)
C. virginica	NB (Canada)	2	Longlines				(Comeau *et al.* 2006)

Table 6 - Documented biomass of oysters and macrofauna at natural oyster reefs

Author	Location	Oyster densities (approx # ·m^{-2})	Oyster size (cm)	No. species associated macrofauna	Total macrofauna (# · m^{-2})*	Total macrofauna biomass (g · m^{-2})**
Dame *et al.* 1984; Dame 1979	South Carolina	1,000 – 2,000		37	2,476-4,077	214
Bahr & Lanier 1981	Georgia	4,000	6+	42	38,000	705
Lehman 1974 cited in Bahr and Lanier 1981	Florida	3,800	All sizes	31	6,200	253
DeAlteris 1988	Virginia	10 -1,000	5-7			
Harris 2003	Chesapeake Bay (constructed)	500 -1,000	Spat on shell	18		
Milewski & Chapman 2002	Caraquet N.B.	67 - 84	All sizes	3 - 14	32 - 216	
ibid	Miramichi N.B.	16 - 164	All sizes	15 - 25	360 – 2,572	
ibid	Cocagne, N.B.	35 - 379	All sizes	18 - 27	440 – 2,848	
ibid	Bouctouche, N.B.	60 – 1,603	All sizes	19 - 29	504 – 6,448	
Sephton & *Bryan 1989*	Caraquet N.B.	250 - 420	All sizes			

* Min-Max reported, **soft tissue wet weight

4.2.4 Potential co-occurrence

Another element to consider is the potential for co-occurrence between the activity and the environment (i.e. competition for space). A common proxy to help assess the potential impact of aquaculture operations is to estimate the proportion of the total bay surface which is occupied by leases. Shellfish aquaculture lease sizes in the Maritime Provinces are mostly small, averages range from 3.51 ha to 15.71 ha (Table 7), but some leases can be considerably larger. In N.B. approximately 60% of oyster leases are smaller than 4 ha (Figure 6). Of the total number of lease sites not registered as vacant, an unknown number of sites essentially lie fallow with little or no effective activity (C. Noris, pers. com.).

Table 7 - Number and surface area of active leases issued by area in the southern Gulf of St. Lawrence, 2001-2002.

	Leases	Total surface area of leases	Average surface area per lease
AREA	Number	Hectares	Hectares
Prince Edward Island	776	2,721	3.51
Eastern New Brunswick	624	2,513	4.03
Gulf Nova Scotia	33	518	15.71
TOTAL	1,433	5,752	4.01

Includes all estuaries in N.B. where oyster aquaculture is conducted, except Baie des Chaleurs.

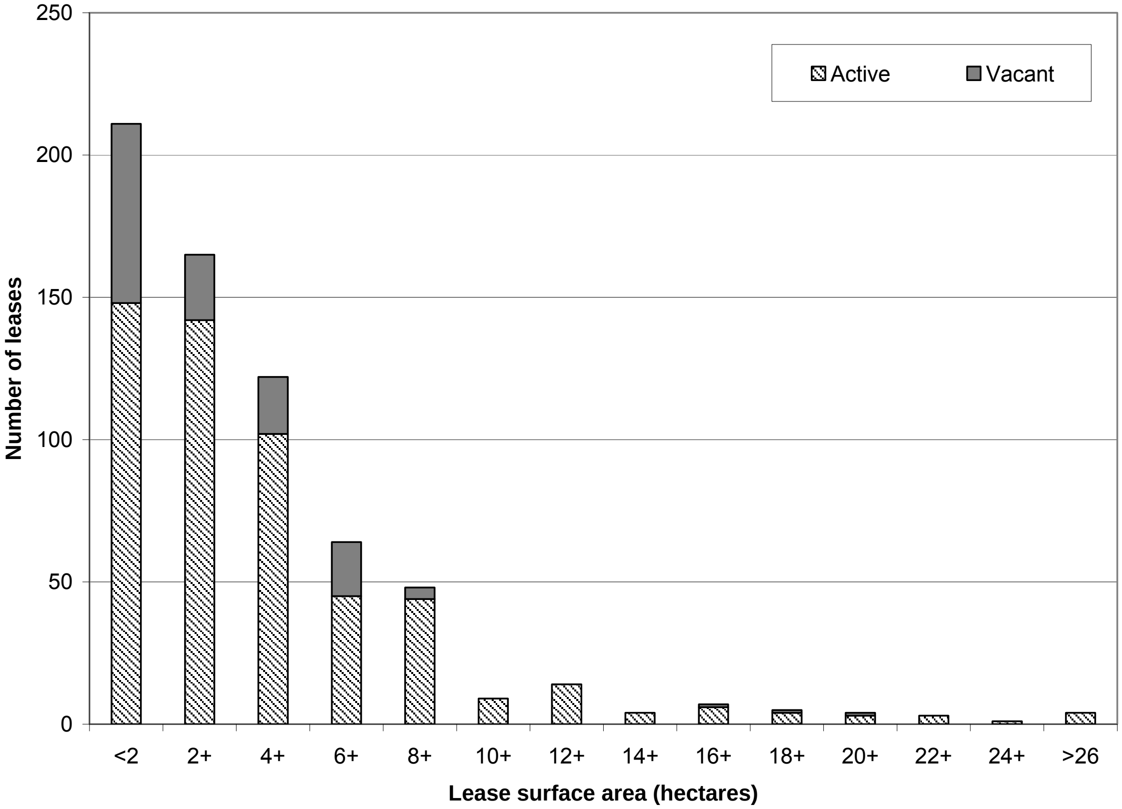

Figure 6 - Surface areas covered by active and vacant oyster leases in N.B. (data from NBDAA, 2007; n=658)

A GIS analysis of the area of leases area to total estuarine waters surface area shows that less than 5% of the surface of N.B. estuaries is defined as lease area, for all techniques included (data from NBDAA). Within these leases, the effective coverage or actual footprint of the aquaculture gear is limited by several factors (e.g. vacant space, navigation channels, unsuitable water depths in the lease, etc.). Thus the effective coverage is calculated as follows:

For a lease with a total surface area of 1 hectare: 100 m X 100 m = 10,000 m^2 (x)

- Longlines -10 per hectare
  - Total surface area occupied by longlines = (60 m X 2.0 m) X 10 = 1,200 m^2 (y)
  - Percentage of lease covered by gear = (y ÷ x) X 100 = 12%
- Cages -240 per hectare
  - Total surface area occupied by cages = (2 m^2 X 240) = 480 m^2 (y)
  - Percentage of lease covered by gear = (y ÷ x) X 100 = 4.8%

Therefore the aquaculture gear typically occupies between 5 to 15% of the surface area of a lease; or less than 1% of most bays. The footprint associated to the gear should likely be considered more representative of the affected area, in terms of fish habitat, rather than the total surface of the lease of which much of the lease area is not utilized.

4.3 Sensitivity of fish habitat

4.3.1 Characteristics of estuaries in N.B.

The biological composition of fish habitats in estuaries is generally found to be dynamic, constantly evolving and responsive to varying environmental gradients (Attrill & Rundle 2002; Attrill & Power 2004). In general, estuaries in eastern N.B. share similar characteristics. They are partially enclosed and protected from the open sea by systems of dunes and barrier islands. The different combinations of freshwater and saltwater inflow, precipitation, temperature, tidal range, dissolved oxygen, sediments loading and wave action lead to the development of a range of connected fish habitats within the estuary. Spatial delimitation between these various fish habitats is defined by nuances in physical, chemical and biological characteristics. These in turn affect the sediment characteristics, nutrient and oxygen availability, desiccation and immersion profiles, water temperature and salinity, etc. Current flow and wave action generally determine the sorting of sands, gravels and silts and the formation of mud and sand flat areas, salt marshes, sand or gravel beaches, shallow inlets and bays.

Eastern N.B. estuaries are generally shallow; as a result, the seasonal range in surface water temperature is among the highest in Atlantic Canada. Typically, water temperatures will reach 16-22°C in the summer; and -1°C to 5°C in the winter (DFO 1996). Seasonal ice generally covers these estuaries between December and March. Hence, overlap of boreal and temperate fish species can be observed with a north-south gradient in species composition.

The sGSL is considered a biologically diverse region and an important spawning ground and nursery area for a number of commercially important fish species (DFO 2001). N.B.'s estuaries contribute significantly to the overall ecological richness and productivity of this region. Two characteristics of this production is the large seasonal increase in plankton and the variety and abundance of larval fish and invertebrate species (DFO 2001).

Intertidal and subtidal areas support pelagic, benthic and burrowing communities of organisms. The location and composition of these communities is determined mainly by the suite of physico-chemical variables. Plant and animal communities depend on ambient conditions for providing nutrients, oxygen and carbon supplies. Another factor influencing the nature of these communities is the bathymetry or depth profile and the degree of wave action. Wave action, particularly during storms, ice-scouring (Robertson and Mann 1984) and exposure may in turn affect intertidal communities. This is likely to be more observable in shallow waters and can result in varying levels of sediment and biota transport and turbidity.

The salinity structure in estuaries is primarily determined by the seasonal freshwater discharge. Attrill and Rundle (2002) suggested that estuarine compartments are mainly defined by salinity which is a primary factor affecting the distribution of estuarine communities. Stable groupings of species tend to follow thermal or salinity boundaries (Attrill & Rundle 2002). In eastern N.B., the salinity gradient typically increases from low levels near the inshore freshwater source to higher levels where the estuary opens into marine conditions of the sGSL. Salinity stratification may occur in deeper regions of the estuary during certain seasons, but typically the shallow periphery of the estuary is homogenous because of active wave and current mixing. Relatively stable salinities are found near the freshwater tributaries and the estuary mouth.

4.3.2 Sensitivity characterization

The principal distinction between natural oyster populations and oyster aquaculture with regards to its influence on the sensitivity of fish habitat is tied to the presence of physical structures which have the potential to have localized affects on the physical characteristics of estuaries, such as wave and tidal currents, turbidity and sediment mixing. When compared to storm events, the influence of physical structures in the water appears minimal, and unlikely to affect the sensitivity of fish habitat. Stochastic natural events are more likely to have significant and widespread impacts on estuarine plants and animal communities than aquaculture activities (Mallet *et al.* 2006). These natural disturbances are believed to be necessary conditions for the maintenance of stable biotic communities, since they promote the redistribution of resources within the ecosystem (Rykiel 1985).

As seen above, variability is inherent to the physico-chemical and biological characteristics of estuaries. Estuarine plant and animal communities need to be able to endure significant seasonal and geographic variability in conditions. They have to be well adapted to survive the physical stresses imposed by these extremes.

4.3.3 Sensitivity of fish species

Many species in the sGSL region are dependent on estuaries for at least a phase of their life history as feeding, nursery, migration and/or spawning habitat. They are thus potentially vulnerable to impacts from habitat alteration. Particularly susceptible are species or species groups that require estuaries or freshwater tributaries as primary larval or post-larval habitat. In the N.B. region, these species include anadromous fish such as striped bass, blueback herring, alewife, American shad, sturgeons, rainbow smelt, Atlantic tomcod and winter flounder.

Other commercial fish species found in estuaries include various species of bivalves, such as mussels, quahogs and clams and crustaceans, such as rock crab and lobster. The effect of water column oyster aquaculture on these fish species is generally considered minimal as the structures do not impede fish movement. In addition, Powers *et al.* (2007) and DeAlteris (2004) demonstrate that aquaculture structures can provide biogenically structured habitats that function as nursery and feeding habitats for juvenile fishes and mobile invertebrates.

4.3.4 Sensitivity of submerged aquatic vegetation

In N.B. the fish habitat most likely to co-occur and to be affected by oyster aquaculture is eelgrass (*Zostera marina*) beds. *Zostera* is considered important in maintaining desirable ecological properties of estuaries due to photosynthetic activity, its role in biomass accumulation and in nutrient cycling. In addition, eelgrass plays a important role as a nursery habitat for a variety of fish and invertebrate species (Locke & Hanson 2004) such as juvenile white hake and small cunners (Joseph-Haché *et al.* 2006).

Several factors are known to affect potential eelgrass growth and recovery (UK Marine special areas of conservation 2006), such as: the removal of habitat; the creation of unstable substrata; the fragmentation and destabilization of *Zostera* beds caused by factors such as changes to coastal processes; physical damage or stochastic weather events; reduced rhizome growth and seed production; reduced light penetration caused by increased turbidity, changes in salinity, pollution or epiphyte smothering; nutrient enrichment; declines in epiphyte grazer populations; unusual increases in waterfowl grazing pressure; non-native macrophyte species, exposure to extreme temperatures, which may increase the susceptibility to disease.

Worldwide, two of the most important threats to submerged aquatic vegetation are disease and anthropogenically induced eutrophication (Short *et al.* 2001). Nutrient pollution effects on eelgrass and nitrogen loading from a variety of sources such as agriculture run-off, sewage, and fish plants are described to varying degrees in N.B. estuaries (Conservation Council of New Brunswick 2004, Lozte *et al.* 2004).

In oyster aquaculture, eelgrass may be affected principally by incidental removal (mooring, boat wash, trampling, etc.), by biodeposition or by shading. This effect is variable in spatial distribution and severity and appears tied to the equipment's footprint. Table 8 describes some impacts of different activities on *Zostera* populations and observations about its resilience. It shows that in general, *Zostera* is not overly sensitive to changes and perturbations. Auster & Langton (1998) observe a consistent pattern of resilience of *Zostera* populations in studies of the impacts of fishing activities. Table 8 also lists pre and post impacts from a number of activities, such as oil spill, herbicide application and wildlife grazing. Other than those cases of intense removal of stems and meristems, effects on *Zostera* appear to have minimal long term impacts.

At present, rarity is generally not a concern in N.B., as *Zostera* meadows are ubiquitous throughout the region and eelgrass is the dominant attached vegetation in these estuaries (Joseph-Haché *et al.* 2006). There are signs, however, that cumulative human activities are having growing impact on these meadows. Increased shoreline developments, recreational and touristic activities are having notable physical impacts.

Globally, studies show that increased nutrient loading to estuaries can lead to eelgrass disappearance (Hauxwell *et al.* 2001,Lotze *et al.* 2003, Cardoso *et al.* 2004). Locke (2005) has observed that the above-ground biomass and percent cover of eelgrass in estuaries along the Northumberland Strait are showing signs of decline; disturbance by the introduced green crab and global environmental changes are mentioned as possible explanations (Locke 2005). Thom *et al.* (2003) suggest that climate variations can have profound effects on eelgrass. They found that large-scale changes climate may strongly influence eelgrass abundance that can vary by as much as 700% annually.

Table 8 - Summary of findings on Zostera sp. recovery and sensitivity to various impacts.

Habitat	Source of effect	Location	Results	References
Eelgrass	Scallop dredge	North Carolina	Comparison of reference quadrats with treatments of 15 and 30 dredging in hard sand and soft mud substrates within eelgrass meadows. Eelgrass biomass was significantly greater in hard sand than soft mud sites. Increased dredging resulted in significant reductions in eelgrass biomass and number of shoots.	Fonesca *et al.* 1984 in Auster & Langton 1998
Eelgrass and shoalgrass	Clam rake and “clam kicking”	North Carolina	Comparison of effect of two fishing methods. Raking and “light” clam kicking treatments, biomass of seagrass was reduced approximately 25% below reference sites but recovered within 1 year. In “intense” clam kicking treatments, biomass of seagrass declined approximately 65% below reference sites. Recovery did not begin until more than 2 years after impact and biomass was still 35% below the level predicted from controls.	Peterson *et al.* 1987 in Auster & Langton 1998
Eelgrass and shoalgrass	Clam rakes (pea digger and bull rake)	North Carolina	Compared impacts of two clam rake types on removal of seagrass biomass. The bull rake removed 89% of shoots and 83% of roots and rhizomes in a completely raked 1 m2 area. The pea digger removed 55% of shoots and 37% of roots and rhizomes.	Peterson *et al.* 1987 in Auster & Langton 1998
Seagrass	Trawl	Western Mediterranean	Noted loss of *Posidonia* meadows due to trawling; 45% of study area. Monitored recovery of the meadows after installing artificial reefs to stop trawling. After 3 years plant density increased by a factor of 6.	Guillen *et al.* 1994 in Auster & Langton 1998

Eelgrass	Recreational clam harvesting	Oregon	Experimentally tested by raking or digging for clams in 1 m2 plots in eelgrass meadows. After three monthly treatments, eelgrass measures of biomass, primary production (leaf elongation), and percent cover were compared between experimental and control (undisturbed) plots. Clam digging reduced eelgrass cover, above-ground biomass and below-ground biomass in measurements made 1 month after the last of three monthly treatments. 10 months after the last clam digging treatment, differences between treatment and control were not statistically significant.	Boese 2002
Eelgrass	Physical exposure	Danish sites	Shallow eelgrass populations form characteristic landscapes with a configuration that is highly related to the level of physical exposure; the size and position of eelgrass beds changes substantially among years	Frederiksen *et al.* 2004
Eelgrass	Experimental removal	San Francisco Bay + Puget Sound	Eelgrass was removed from experimental plots. Substantial vegetative recolonization (64.3 -81.8%) of test plots occurred within five months of treatment. Rapid recolonization was explained by the presence of new shoots migrating to excavated plots and reseeding.	Fonsecal *et al.* 1983
Eelgrass	Mussel dragging	Maine	Aerial photography, underwater video, and eelgrass population- and shoot-based measurements were used to quantify dragging impacts within 4 sites that had been disturbed at different times over an approximate 7 year interval, and to project eelgrass meadow recovery rates. Dragging had disturbed 10% of the eelgrass cover. Dragging removed above- and belowground plant material from the majority of the bottom in the disturbed sites. One year following dragging, eelgrass shoot density, shoot height and total biomass of disturbed sites averaged respectively 2 to 3%, 46 to 61% and <1% relative to the reference sites. Substantial differences in eelgrass biomass persisted between disturbed and reference sites up to 7 year after dragging. The pattern and rate of eelgrass bed recovery depended strongly on initial dragging intensity; areas of relatively light dragging with many remnant eelgrass patches (i.e. patches that were missed by the mussel dredge) showed considerable revegetation after 1 year.	Neckles *et al.* 2007

Eelgrass	Canada geese grazing	Maine	A flock of Canada geese Branta canadensis L. over-wintered and grazed on eelgrass for 3 months. Before Canada geese were present, eelgrass parameters demonstrated seasonal fluctuations typical of the region. During the grazing event, eelgrass parameters declined drastically, and biomass losses were significant. After the event, eelgrass recruitment via sexual reproduction was minimal, and vegetative recovery was impeded by Canada goose consumption of the plant meristems. Unlike studies in other locations, which show seagrass quickly rebounding from annual grazing events, eelgrass in this location showed little recovery from grazing 1 year after the event.	Rivers & Short 2007
Eelgrass	Wasting disease	Delaware USA	Eelgrass declined precipitously in the 1930s due to the pandemic wasting disease and a destructive hurricane in 1933. Natural recovery of *Z. marina*, possibly deriving from either small remnant stands or undocumented transplant projects was significant in four northern bays, with over 7319 ha reported through 2003 compared to 2129 ha in 1986, an average expansion rate of 305 per year. This rapid spread was likely due to seeds and seed dispersal from recovering beds.	Orth *et al.* 2006
Zostera sp.	Exposure to herbicides Atrazine, Diuron and Irgarol	Laboratory & Australia	*Zostera capricorni* was exposed to 10 and 100 µg herbicide solutions for 10h. Laboratory samples exposed to these herbicides were severely impacted during the exposure period and most treatments did not recover fully after 4 days. In situ samples were severely impacted by Irgarol and Diuron exposure whereas samples recovered completely after exposure to Atrazine at the same concentrations as the laboratory experiments.	MacInnis & Ralph 2003
Zostera sp.	Brant goose grazing	Europe	”Wasting disease" affecting Atlantic Zostera stocks during the 1930s was at least partly responsible for a steep decline in Brant goose population sizes on both sides of the Atlantic. While *Zostera* is of outstanding importance as food for Brant geese, the impact of the geese on *Zostera* stocks seems to be less important - at many sites, the geese consume only a small amount of the available *Zostera*, or, if they consume more, the seagrass can regenerate fully until the following season.	Ganter 2007

Eelgrass	Oil spill	Alaska	A year after the Exxon Valdez oil spill, eelgrass densities were 24% lower at oiled sites compared to control sites. Recovery of eelgrass occurred by the second year, with no significant differences noted between oiled and control sites in subsequent years.	Dean & Jewett 2001
Zostera sp.	2-4-D herbicide	New Brunswick	The industrial herbicide 2-4-D was used to clear eelgrass from oyster grounds in part of Baie Brulée in 1968. Surveys in 1986 showed that the area was densely vegetated with eelgrass; eelgrass beds covered 97.7% and 46.1% of the area of the bay in St. Simon Sud and St. Simon Nord, respectively.	*Mallet pers. com.* SEnPAq 1990a
Zostera sp.	Oyster aquaculture	California	Study plots were established to test the effect of oyster line spacing distances of 1.5 ft (narrow), 2.5 ft (standard), 5 ft (wide) and 10 ft (very wide). They examined the eelgrass, benthic infauna cores, deployed baited fish traps and measured water quality, sedimentation, light intensity, and oyster growth. After two years, eelgrass spatial cover and shoot density were consistently high within the control (reference areas) and lowest within the 1.5 ft oyster line spacing plot. Eelgrass metrics generally scaled directly with oyster density, and the spatial cover and density of eelgrass plants within the 10 ft spacing plot were within the range of variability observed in the reference (control) study plots.	Rumrill & Poulton 2004

4.4 Significance of ecological risk

The concept of significance can not be separated from the concepts of "adversity" and "likelihood" and must be considered by taking into account the implementation of mitigation measures (CEAA 1994). The following definitions represent guidance for the determination of significance and were elaborated based on the CEAA and the HMP Risk assessment framework:

- *Significant* : A residual environmental effect is considered significant when it induces frequent, major levels of disturbance and/or when the effects last longer than a year and extend beyond the project boundary following the application of mitigation measures. It is either reversible with active management over an extended term or irreversible.
- *Not-significant* : A residual environmental effect is considered not significant when it is infrequent, minor or negligible levels of disturbance and/or damage and when the effects last less than a year and are contained within the project boundary following the application of mitigation measures. An effect that is not significant is reversible with or without short-term active management.

The significance of the ecological risks associated with water column oyster aquaculture is based on the best current available information in the context of our understanding of the ecosystem dynamics. The following points discuss some of the more complex concerns that are typically raised and where ongoing research occurs in regards to water column oyster aquaculture effects on fish and fish habitat.

4.4.1 Biodeposition

One of the principal concerns with regards to the potential negative effects of bivalve culture is related to the increased deposition of organic matter associated with the accumulation of faeces and pseudofaeces as well as the deposition of shells and attached epifauna from the structures and changes to the hydrodynamics of the site. The impact of these effects on the benthos can range from significant, in the case of intensive Asian and European culture practices, to minimal in the case of semi- to low-intensity operations; (Chamberlain *et al.* 2001, Crawford 2003, McKindsey *et al.* 2006a). It would therefore appear that there is a potential for localized negative effects on the ecosystem due to increased organic loading within the footprint of individual farms under certain conditions (e.g. heavy loading, low flushing rates, shallow water depth, etc.).

Models can be used to predict the dispersion of biodeposits as they fall from the aquaculture works and assess the extent of the activity's footprint. Chamberlain et al. (2006) show that in shallow depth sites, such as in water column oyster aquaculture, deposits are expected to fall largely under the equipment. They also show that particle flux is correlated to the stocking density of the cultured species and also that resuspension and mixing of these particles are likely to occur in shallow systems. Thus, the impact of biodeposition depends largely on the density of shellfish present at the site and extent to which water exchange will disperse of the deposits.

In the case of water column oyster aquaculture, studies on sedimentation rates in St. Simon Bay N.B. showed that deposition rates increased at culture sites possibly from the oysters, fouling organisms and hydrodynamic effects of the equipment (Mallet *et al.* 2006). However, the mean organic content of the sediment deposited at the Oyster Table site (20.2%) was not significantly different from the Floating Bag (20.8%) or the Reference sites (21.8%) (Mallet *et al.* 2006). The authors suggested that the lack of enrichment of the sediments indicated that the organic matter in the biodeposits was not being incorporated into the sediments and was either washed away and/or rapidly processed by the benthic community.

When organic enrichment occurs, as seen in intense finfish aquaculture, it can cause alterations in the benthic community; reducing species diversity and richness as the impact accentuates (Pearson and Rosenberg 1978; Rosenberg 2001). Mallet *et al.* (2006) concluded that, the number of species and macrofaunal abundance was similar at the culture and the reference sites, and there was no evidence of opportunistic species typically associated to highly disturbed areas.

The use of Eh/Sulphide analyses of the sediments was developed for finfish aquaculture as a quantitatively index of organic enrichment and the formation of anoxic sediments and levels were correlated with the composition of the benthic community (Wildish *et al.* 2001). This technique has been applied elsewhere but to date no significant impact was detected for the analyses of the sediments under oyster sites (Mallet *et al.* 2006) in Baie St. Simon N.B., one of the most intensively cultured bays in the Province. Mallet *et al.* (2006) found that Eh/Sulphide levels at oyster sites were not significantly different from reference sites. Additionally, as part of an MOU with the N.B. provincial government, the NBDAA has initiated surveys to measure

Eh/Sulphide levels in and around oyster aquaculture sites. In 2006, sites within two bays which are considered important oyster aquaculture areas were assessed on and off leases. They found that in Baie St. Simon and in Richibucto, levels of sulphides in the sediments averaged 314 µM and 159 µM, respectively (data from NBDAA). The maximum levels observed 1410 µM and 1165 µM for Baie St. Simon and Richibucto respectively, occurred outside the leases in the deeper areas of the navigation channels (data from NBDAA). Hypoxic conditions in the sediments occur at sulphide values of 1,500-3,000 µM while anoxic conditions correspond with levels of 3,000-6,000 µM or more (e.g. Wildish *et al.* 2001, Holmer *et al.* 2005).

Therefore, there is no indication to date of significant or adverse effects associated with the increase in biodeposition under water column oyster aquaculture sites in N.B.

4.4.2 Carrying capacity

There is concern over to the potential effect of increasing the oyster biomass on the carrying capacity of estuaries. As shown in section 4.2, the intensity observed in water column oyster aquaculture in N.B. differs significantly from other regions in the world.

Several attempts have been made internationally to determine the carrying capacity of estuaries for shellfish production. One of the main obstacles is the lack of clarity in the definition of carrying capacity. For shellfish culture, McKindsey *et al.* (2006b) favour the use of “ecological carrying capacity” which represents the point where the stocking density on the farm is high enough that it can cause *unacceptable environmental impacts*. Typically, the carrying capacity for shellfish is based on the biomass which can be supported in a given bay in terms of food, habitat, water quality and other necessary parameters. Research in this area has been limited by the complexity of seasonal and size related changes in energy requirements of shellfish, seasonal changes in productivity, trophic characteristics of estuarine communities and hydrodynamics of many areas. Various problems have been reported in the literature about models used to determine carrying capacity and their requirement for long term environmental data collection. Newell (2007) highlights the shortcomings of current models in accurately representing conditions observed in shellfish aquaculture and lists the steps required to improve these efforts; including a better account for ecosystem functions provided by bivalves which have desirable (e.g. economic, environmental remediation and nutrient trading scheme) outcomes. In particular, these models need to take into consideration the cumulative effects of neighbouring human activities (e.g. nutrient run-off, sedimentation, etc.) (ICES 2003).

The carrying capacity of a given system is not at a static or unchanging level. Seasonal changes in temperature, food supply or other factors can affect the capacity of a bay or estuary to support the organisms within it (Carver and Mallet 1990). Bivalve culture is strongly influenced be the quantity of food (i.e. plankton and organic particles) which is available in the water column. The Aquatic Ecosystem Section of DFO in the Gulf region initiated the Shellfish Monitoring Network (DFO 2007) (https://www.glf.dfo-mpo.gc.ca/sci-sci/smn-rmm/intro-e.jsp) in order to examine spatial and temporal variations in shellfish productivity using standardized cage systems in bays with oyster or mussel culture in the sGSL. For example, differences in growth rates of bivalves in different bays between years are often more important than differences between bays despite varying intensities of bivalve culture within the bays. This suggests that productivity is linked more strongly to broad annual changes in nutrient inputs, plankton blooms or temperatures than to grower interventions within a given bay. This monitoring of shellfish productivity is ongoing and will continue to provide a baseline of shellfish growth so as to provide an indication of ecosystem effects if changes outside of the natural variability are observed.

Given historical levels of natural oysters within N.B. estuaries (see section 5.2 *Historical state of oyster populations*) as well as the comparisons with bivalve production in other regions of the world, it appears that the ecological carrying capacity of these systems is not likely to be adversely affected by the anticipated level of water column oyster aquaculture.

4.4.3 Nutrients

The effect of nutrient releases such as nitrogen and phosphorous from farmed oysters in the form of faeces and pseudofaeces is generally considered of lower importance compared to the regional inflow of nutrients in open water masses (Folke & Kautsky 1989, Kirby & Miller 2005, Ferreira *et al.* 2007). Generally, the excretions that oysters do produce are thought to be rapidly assimilated by plankton in the water column (Pietros and Rice 2003). Shellfish in culture consume ambient plankton and are not artificially fed. Thus they do not add nutrients but rather can alter the nutrient dynamics and concentrate nutrients in the farm's immediate surroundings (McKindsey *et al.* 2006a). This concentration of nutrients can be difficult to assess in the water column and explains why appreciable efforts are made to study benthic enrichment and biodeposition, as discussed above.

Unlike finfish aquaculture, where one of the main ecosystem stressors is related to the addition of nutrients, chemicals and pharmaceuticals in the form of fish food, bivalve aquaculture represents an extractive activity, by which the bivalves filter food out of the water column and these nutrients are removed from the ecosystem entirely at harvest. Sarà (2006) conducted a meta-analysis on the ecological effects of aquaculture on nutrients by comparing shrimp, fish, bivalve culture as well as polyculture. The author concluded that the effect of aquaculture on nutrients was highest in freshwater and lowest in marine water. Moreover, the author found that bivalves appeared to have no significant influence on the dissolved nutrients and their “mean size of effect” was negative (-0.03) unlike the positive values seen in shrimp (+0.71), fish (+1.10) and polyculture (+1.80) (Sarà 2006).

That said, although oysters are known to have been highly abundant historically, the role of shellfish aquaculture in influencing the nutrient dynamics in estuaries, as well as in selective grazing of plankton, remains an ongoing research topic.

4.4.4 Submerged aquatic vegetation

Another common concern relates to the potential damage to submerged aquatic vegetation which is considered valuable habitat for several fish species (e.g. Chambers *et al.* 1999; Joseph *et al.* 2006; Vandermeulen *et al.* 2006). Marine plants such as eelgrass are considered critical habitat in many parts of the world because they serve important ecological functions, are often considered rare, and thus are often the subject of monitoring programs (Short *et al.* 2001). It is important for many fish and invertebrates and contributes to the ecological richness of the region. In N.B. estuaries, the eelgrass (*Zostera marina*) is considered abundant in many bays. Surveys have shown that eelgrass beds can represent appreciable portions of N.B. estuaries (SEnPAq 1990ab). For example, in Baie St. Simon Sud and in Richibucto Bay 98% and 78% of the surface area of the bays, respectively, was covered by eelgrass beds; these values do not exclude sediment types unsuitable for eelgrass. The SEnPAq (1990ab) study is currently being used as a baseline with which to compare eelgrass distribution. A DFO working group is presently assessing eelgrass as a potential indicator for evaluating bay health in N.B. in collaboration with Environment Canada, Parks Canada and universities.

Eutrophication remains the main concern to eelgrass productivity and is recognized as a threat by increasing epiphytes on the leaves, and reducing water clarity which cause shifts in the primary productivity from benthic vegetation towards phytoplankton. It is clear from the scientific literature that shellfish filtration plays a critical role in improving water clarity which increases light availability and enhances bioavailability of nutrients and thereby stimulating eelgrass growth (e.g. Kennedy V.S. 1996; Newell & Koch 2004; Kirby & Miller 2005; Newell *et al.* 2005). This positive interaction can apparently be reduced in certain highly eutrophic settings such as in the Thau Lagoon in France (e.g. DeCasabianca et al. 1997 and 2003).

Other concerns may relate to physical disruptions as eelgrass can be dislodged by aquaculture activities such as trampling, anchoring, and powerboat wash. Past practices, whereby oyster culture was conducted by partial removing of eelgrass in order to facilitate removal of oysters and increase water flow, are no longer carried out. Vandermeulen *et al.* (2006) state that the preservation of habitat can be achieved by ensuring adequate spacing between lines and by minimizing physical impacts. Rumrill and Poulton (2004) found that oyster aquaculture gear placed line-spacing at 3m exhibited eelgrass metrics that fell within the range of variation observed in a series of reference areas while significant impacts occurred at smaller line spacing. The current space left for boat navigation (typically >7 m) is typically greater than the (>3 m) minimum spacing which was recommended by the NAP (Vandermeulen *et al.* 2006). Dumbauld (2005 cited in Vandermeulen *et al.* 2006) states that eelgrass can recover in 1-2 years if left undisturbed.

Stephan *et al.* (2000) compiled results on the effects of impacts of fishing gear (i.e. dredging, trawling, raking, etc.) on submerged aquatic vegetation and qualified the “injury recovery potential” of eelgrass Zostera marina as “moderate” in comparison to ten other species marine vegetation. Peterson et al. (1987) evaluated the effect of different intensities of mechanical harvesting of clams (*Mercenaria mercenaria*) including the “clam-kicking” technique which involves directing the propeller wash downward with sufficient force into the sediment to displace the sediments thus exposing the clams for easy collection with a trawl. They found that “intense kicking” had significant effects in reducing eelgrass biomass while “light-kicking” and raking had much lower impacts. Eelgrass in the “light-kicking” and raking treatments recovered to the level of the controls within 1-year.

Based on the studies of eelgrass resilience to anthropogenic activities presented above and natural disruptions (e.g. grazing, ice-scours, annual variability with environmental conditions), the potential effect of these physical disruptions associated to water column oyster aquaculture is likely to affect a limited area and to be fully reversible.

4.4.5 Species interactions

Concerns with regards to species interactions typically relate to the presence of additional oysters in the water column. Abgrall *et al.* (in prep.) completed a review of intra and inter-specific interactions between the oyster and softshell clam (*Mya arenaria*). Although cultured and wild population interactions, such as predation, competition, etc. are likely to occur, there is no indication that these interactions differ significantly from those occurring between wild populations. Oysters cultured in the sGSL are native to this region and have co-existed with other native species; therefore they are expected to retain similar biological interactions with existing populations.

Other concerns relate to the structures in the water column. The study of the effects of these types of structures has evolved into a research field which refers to them as Fish Aggregation Devices (FAD) (e.g. Castro *et al.* 2002). Some authors have proposed that the aquaculture equipment itself, and other structures, may contribute to estuarine productivity by creating a hard substrate; availability of these surface areas can limit the colonization of certain organisms (McKindsey *et al.* 2006a). Passing from an essentially two-dimensional sand-mud habitat to a three-dimensional hard surfaced habitat can dramatically alter the surface area available.

DeAlteris *et al.* (2004) conducted a study to compare the relative habitat value of aquaculture gear (rack and bag), submerged aquatic vegetation (*Zostera marina*), and shallow non-vegetated seabed. They found that the ecological value of aquaculture gear was significant based on an assessment of resident and transient marine organism’s abundance and diversity in the respective habitats. Aquaculture gear increased habitat complexity and supported higher abundances of organisms than non-vegetated seabed; this was determined to be particularly beneficial to recreational and commercial fish and invertebrate species in their early life stages. DeAlteris *et al.* (2004) concluded that the relative habitat value of aquaculture gear is at least equivalent to submerged aquatic vegetation. Powers *et al.* (2007) demonstrated that flora and fauna associated to clam aquaculture gear (netting) was significant and that community structure of mobile invertebrates and juvenile fishes utilizing leases was more similar to that of seagrass than sandflat habitats. They found that the utilization by juvenile fishes was 3 times greater in seagrass and 3 to 7 times greater in epibiota on mesh in clam leases than on sandflat habitat.

Similarly, a study done in the sGSL in 2006 monitored levels and types of epifauna found on floating oyster bags (Mallet *et al. in preparation*). Undisturbed oyster aquaculture bags can accumulate 500 g to 1500 g (wet weight) of epifauna (e.g. amphipods, algae, arthropods, molluscs, etc.) per bag in one season. This can have important ramifications for the food web. For example, the estimated abundance of the tube amphipod *Jassa* sp. reached over 185,000 individuals per bag in the fall. This may represent an abundant food source for small fish (e.g. sticklebacks, silversides, cunner, etc) which appeared to be feeding on the surface of the bags (Mallet *et al. in preparation*).

In addition, the epibenthic fauna community was assessed in areas of suspended bag oyster aquaculture in three N.B. bays in 2006. Trawls were collected within leases (0 m) and at subsequent distances of 25 m, 100 m and 500 m away from lease edge (Skinner *et al.* in preparation). In general, it was found that the total organism abundance and species richness was significantly higher at lease sites than off-lease; lease site communities were generally dominated by shrimp species and blue mussels. The contribution of aquaculture gear to habitat value is explained in part by the fact that oyster culture creates several compartments (hard and soft substrata, foraging, refuge and nursery habitat) and trophic levels (primary producers, filter-feeders, deposit-feeders) within the water column (Mazouni *et al.* 2001). However, opportunistic predators such as sea stars and rock crab (*Cancer irroratus*), which can be abundant in mussel aquaculture sites and seen feeding on mussel fall-off, were only observed infrequently at the oyster aquaculture sites (Hardy, unpublished data).

4.5 Conclusion of Ecological Risk Assessment

This Ecological Risk Assessment identifies and characterizes many of the risks to fish and fish habitat relating to water column oyster aquaculture and discusses them in the context of the scientific literature and ecosystem dynamics. It is important to note that this assessment should be considered by habitat managers as a starting point and be revisited as new information becomes available.

The research priorities identified in the NAP as well as others, once completed, will further enhance and clarify some of the uncertainties involved with this activity. Moreover, we recognize that uncertainties exist and will continue to exist as these are complex ecosystems and more scientific research in this field is encouraged.

It is clear that the “scale and intensity” of the shellfish aquaculture operation is the main driver leading to potential negative effects. Culture of the native oyster in N.B. is practiced at densities much lower than other regions in the world and the potential effects are considered reversible and generally limited to site footprint. Based on the risk assessment matrix, our view is that the residual “scale of negative effects” associated with water column oyster aquaculture, as practiced in N.B., is low.

In terms of sensitivity, eelgrass beds are the principal driver in the risk matrix as they are considered important but are ubiquitous in many N.B. bays. Eelgrass also appears to be resilient to severe impacts, provided water quality is maintained. Eutrophication and turbidity appear to be the main factors affecting water quality and thus eelgrass sensitivity. Ensuring water quality should likely be the focus for eelgrass health. Because of concerns with water quality in general, our view is that level of “sensitivity of fish and fish habitat”, based on the risk assessment matrix, is moderate.

In our view, the potential residual negative effects associated to this activity can likely be managed with appropriate planning and mitigation measures. Water column oyster aquaculture, as practiced in Gulf N.B., is not considered likely to significantly harm the productive capacity or the integrity of the fish habitat in these ecosystems. Therefore, overall based on the current state of knowledge and the scale of water column oyster aquaculture, we conclude that the potential residual risk for significant adverse impacts on fish and fish habitat to occur is low and that this constitutes a low-risk activity.

This view is also consistent with a DFO’s Aquatic Ecosystem Section advice on water column oyster aquaculture as practiced in Gulf N.B., with a broader view on the role of aquaculture (similar to NEBA considered in the following section). They concluded that this activity represents a low risk to cause negative effects on fish habitat based on:

- *the current husbandry practices* (and the Code of Practice) *employed by the oyster aquaculture industry;*
- *the relatively low biomass of oysters on an aquaculture lease;*
- *the existence of naturally occurring reefs at densities in excess of the biomass used in aquaculture;*
- *the high historical landings of oysters in N.B. which suggests a high natural carrying capacity;*
- *the nature of shellfish as filter feeders in consuming and recycling nutrients;*
- *the problem of increasing nutrient load of estuaries associated with human activities and the ability of filter feeders to help mitigate these effects;*
- *the harvesting of the shellfish on a yearly basis which can remove tonnes of organic and inorganic matter from the bays; and*
- *the culture of oysters over the past decades in N.B. without significant demonstrable adverse effects.*

5 NET ECOLOGICAL BENEFIT ANALYSIS

In the above risk assessment, the potential for environmental impacts of aquaculture works were considered, here we consider the potential remediation role that oysters can play. The NAP concluded that bivalves in culture appear to fill many of the same ecological roles as natural bivalve communities, a role considered generally beneficial for a number of components of temperate estuarine ecosystems.

Although oysters in aquaculture differ from reefs in their structural form, it is useful, in the current assessment, to consider the ecological services played by oysters. Coastal ecosystems and estuaries dominated by bivalves exhibit complex responses that are not easily explained by linear dynamics (Dame *et al.* 2002). Net environmental benefit analysis (NEBA) is an elaboration upon the conclusions of an ecological risk assessment which considers benefits, along with risks, which can help managers in their decisions (US Department of Energy 2003).

5.1 Historical state of oyster populations

Milewski and Chapman (2002) provided a synopsis of the history of oysters in the province as well as their ecological function and the challenges they face. A relatively complete time series of oyster landings spanning between 1876 and today can be reconstructed from published information allowing us to retrace the evolution and trends in landings for the last 130 years. This gives a relatively reliable chronological series for the evolution of the oyster harvesting industry prior to the arrival of aquaculture. Newell (1988) proposed the use of this kind of time-series as a means to infer information about past standing stocks of oyster reefs. Based on Newell's example, data for landings were obtained from a number of sources; from 1876 to 1969 data obtained from Morse (1971); from 1971-1984 data obtained from Jenkins (1987) in imperial pounds was converted to metric tonnes; from 1984 – 2004 data was compiled by DFO from statistics obtained via sales slips, shown in the following graph (Figure 7). This data demonstrates the general trends in the exploitation of natural stock of oysters. It also helps to illustrate the scale of natural populations prior to current harvests.

At their highest in N.B., reported landings reached a peak in the order of 4,000 t, around the end of the 1940's. They had remained between the 1,000 to 1,500 t in the 75 years prior to that. Since then, NB landings have remained consistently below the 500 t mark, with no indication of commercial landings returning to pre-Malpeque numbers (Table 10).

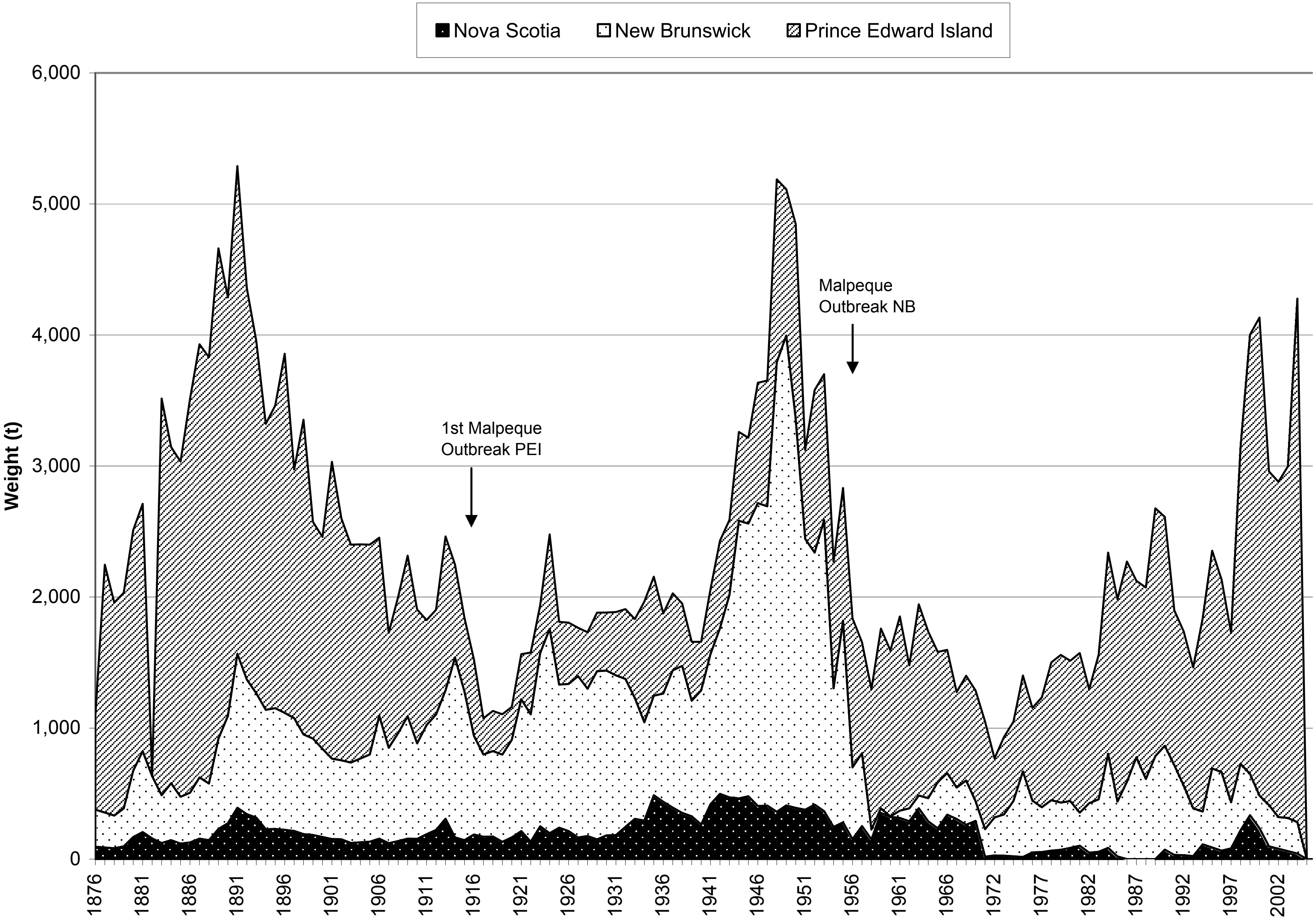

Figure 7 - Reported landings of oysters from commercial harvest 1876-2004 (Morse 1971,Jenkins 1987, DFO 2003b).

Table 9 - Estimated historical quantities of oysters in the Maritime Provinces (based on Newell, 1988) compared to present aquaculture and fishery levels.

	Gulf NS Total Weight (t)	*Gulf NB Total Weight (t)*	*PEI Total Weight (t)*	*Total Total Weight (t)*
1870-1900 estimated oyster biomass	10 161	35 912	130 565	176 638
Estimated total aquaculture production (NBDAA, 2006; DFO statistics)	232	1 857	2 849	4 939
Estimated total aquaculture production –all sizes (Comeau, 2006)		1 249		
Estimated oyster fishery landings 65 mm + (Comeau, 2006)		75		

From these values, and based on Newell's (1988) approach, we can estimate that there would have been a standing stock in the order of 176,000 t of oysters in all three Maritime Provinces prior to the 1900's; 10,161t for N.S; 35,912 t for N.B. and 130,565 t for P.E.I. Considering the fact that landings are generally under-reported and that by the turn of the 20th Century, a number of oyster beds in the Maritimes were already considered depleted (Morse, 1971) it is fair to assume that these numbers would represent a conservative estimate.

Based on the provincial estimates and the Comeau *et al.* (2006) survey, current commercial and aquaculture productions combined would represent less than five percent of the historical biomass of oysters. Therefore, this suggests that the combined standing stock of oysters found in N.B. estuaries is significantly lower than the biomass that would have been observed at the turn of the 20th Century. This is consistent with trends reported elsewhere in the literature (Kirby, 2004).

This historical data of oysters in the Maritime Provinces suggest a high natural carrying capacity and a natural dominance of oysters in these estuarine ecosystems.

5.2 Characterization of reference state

Reference states are typically established based on pre-activity levels (i.e. before the introduction of aquaculture). However, as shown above, the “baseline” by which we typically compare the development of these activities have already shifted drastically from historical levels. Determining where to locate the benchmark for comparisons and assessing what is a “natural and productive ecosystem” is difficult given that our current viewpoint is already far removed from previous levels. The reference state of many estuaries in N.B., as in many areas on the Atlantic coast of North America, was characterized by an abundance of oysters at a level which is now difficult to imagine (Gosling, 2003, Kennedy, 1996). The exercise above of examining historical levels does provide a better perspective for evaluating the scale current changes in our ecosystems and assessing the role of the oyster as a key component to what was presumably a diverse, functional and productive ecosystem.

5.3 Ecological benefit characterization

McKindsey *et al.* (2006) describes effects of shellfish aquaculture on fish habitat. The report provides detailed information on the role of bivalves in the ecosystem under natural conditions, describes various shellfish culture methods, and evaluates whether those roles are mimicked under aquaculture conditions. Their review of literature shows that bivalves are key components of healthy fish habitat.

Moreover, several of authors have argued that oyster reefs can play a critical role in the dynamics and resiliency of temperate estuaries. The reader can refer to the extensive review by Dame (1996): *The ecology of marine bivalves, an ecosystem approach*. They make the argument that oysters and their reefs contribute to the robustness of temperate estuaries; for that reason, they have been termed keystone meta-populations (Dame *et al.* 1984,Ray *et al.* 1997); biogenic habitats (Kennedy V.S. 1996,Lenihan 1999,Cranfield *et al.* 2004,Kirby & Miller 2005); ecosystem engineers (Coen *et al.* 1999,Gutierrez *et al.* 2003); essential fish habitat (DeAlteris *et al.* 2004); and critical estuarine habitats (Coen *et al.* 1997,McCormick-Ray 2005). These ecological roles are summarized in Table 10.

Table 10 - Summary of the functional effects of natural oyster populations on estuarine components (based on Ray et al. 1997; Kennedy V.S. 1996; Ruesink et al. 2005; McCormick-Ray 2005)

ESTUARINE FUNCTION	ECOLOGICAL SERVICES PROVIDED BY OYSTERS
Benthic productivity	Adds nutrients and precipitates faeces and pseudofaeces to benthos to feed demersal feeders, including lobster, crabs and endobenthic organisms. May depress the ratio of centric diatoms (planktonic and eutrophic waters) in favour of pennate diatoms (benthic and clear waters).
Biodiversity	Provides increased niche space for ecological complexity and faunal abundance; supports stenohaline species along a salinity gradient; sustains epizoan diversity; modulate estuarine population structure toward desirable equilibrium. Provide substrate attachment for plants and invertebrates.
Coupling of nutrients to other habitats	Benthic-pelagic coupling of nutrient. Consumption of phytoplankton containing organic nitrogen NH_4^+. Enhances N releases by sediment to atmosphere. NH_4^+ re-uptake by phytoplankton. Enhances composition of nutrient readily available to SAV.
Estuarine resilience and ecosystem robustness	Forms meta-populations and contribute to other communities as sources to restock disturbed areas; long-term life span of oysters contribute to biomass stability in estuaries. Increase habitat heterogeneity in the system and increase habitat redundancy, which can add optional choices in species survival.
Filtration capacity	Permanent presence of long-lived bivalves exerts effective grazing control on phytoplankton. High turnover rate potential of estuarine waters. Preferential sorting of organisms by size, limits impacts on zooplankton; dampens algal blooms; filters bacteria from water column.
Habitat structure	Reefs form discrete hard substrate islands which provide limiting substrate. Shells provide 3D substrate to other organisms for spawning, nursery and refuge habitats. 1 m^2 of shell bottom represents 50 m^2 of surface area for epifauna. These organisms act as food sources for a variety of predators. Reefs provide migration and feeding halts, creates matrix of seascape habitats which connects resource patches to the benthos, marshes and other estuarine habitats. Dead shells can help stabilize benthos, substrate for spat settlement and are recycled over time. Provide refuge from extreme environmental conditions.
Light regime	Removes POM/PIM from water column and enhances depth of Photosynthetically Active Radiation (PAR).
Metabolic conversion	Feeds on phytoplankton and converts energy into secondary production; release gametes and larvae which feed other organisms, including zooplankton and other filter-feeders. Forms spatial nodes of biological activity and couples benthic heterotrophic activity to intense predator-prey interactions. This helps make temperate estuaries among the most productive natural systems known (1 514 $gCm^{-2}y^{-1}$).
Shoreline and sediment processes	Reefs buffer against moderate storms and wave actions. Prevent the erosion of channel banks, stabilize and protect the edges of salt marshes. Mucus-bound biodeposits have enhanced particle cohesion and can resist erosion. Water flow patterns. Alters benthic boundary layer and water column hydrodynamics which enhances particle movements, feeding opportunities and particle dispersions.

5.4 Comparison of alternate states

The critical role of oyster reefs is made the more apparent when they disappear from estuaries, such has been the case in the eastern United States (Kirby 2004). Rothschild *et al.* (1994), for instance, estimated that total oyster habitat in the Maryland portion of Chesapeake Bay is probably 50% or less of what it was a century ago, that the remaining habitat is of substantially poorer quality on average, and that the biomass per unit habitat is about 1% of that at the turn of the century.

Such dramatic reductions in oyster populations are believed to have lead to cascades of undesirable effects on community and ecosystem dynamics, such as the loss of top-down control mechanisms on phytoplankton, which may have resulted in increases in nuisance and toxic algal blooms, reduced water clarity, loss of submerged aquatic vegetation and loss of fish populations (Kennedy V.S. 1996,Kirby & Miller 2005). It is reasonable to assume that a comparable state of reduced contribution of the oysters to estuarine ecology exists in our region, as that historical trend of systematic reef depletion has followed a similar course along the eastern seaboard (Kirby 2004). This would represent a significant loss to the productivity and function of these ecosystems as well as a likely reduction in water quality.

The current state is one of depleted natural oyster populations. It is estimated that populations diminished by more than 90% following the Malpeque disease. In some regions a 100 to 1,000 fold increase in population would be required to restore the desired services provided by oysters (Luckenbach 2004). Bivalve aquaculture is increasingly recognized as being critical in providing important ecosystem services and public benefits, such as mitigating water quality degradation (Powers *et al*., 2007).

5.5 Significance of ecological benefits

The significance of the ecological benefits of oysters can be observed in the decisions to invest a great deal of resources in the restoration and reintroduction of oysters. In particular, the rehabilitation of oyster reefs in temperate estuaries is considered critical in promoting a desirable state of equilibrium, characterized by a production of fish species considered useful to society (Ulanowicz & Tuttle 1992, Peterson *et al.* 2003). They conclude that increasing the number of oysters, naturally or via aquaculture, would result in increased benthic primary productivity, fish stocks, and zooplankton densities.

Bivalve shellfish are increasingly considered for their role in restoration programs and their use in mitigating negative impacts of land use activities (Landry 2002). Over the past years, DFO-HPSD has issued several *Fisheries Act* subsection 35(2) Authorizations on projects located in the estuarine and marine environment. Because these projects (e.g. wharfs, bridges, etc.) were determined to cause harmful alteration, disruption or destruction (HADD) of fish habitat, the proponents were required to compensate for lost fish habitat. In the Gulf Region, most of the marine fish habitat compensation projects are related to reef creation because of their positive ecological functions (Godin pers. com.). Restoration of oyster reefs is typically recommended as compensation to offset the damages to fish habitat in other regions of the world as well, and the net environmental benefits of such interventions are considered positive (Newell 2004, Kirby & Miller 2005, Newell *et al.* 2005). Restoration of natural oyster reefs is recognized as having significant ecological benefit and is often recommended as the preferable option because of the overall gains in habitat structure and function.

In the United States, the National Oceanic & Atmospheric Administration (NOAA) is actively involved and making significant investments in the restoration of oyster populations: in Chesapeake Bay alone, this funding represented 5.4$ million in 2006 (http://chesapeakebay .noaa.gov/RestorationMain.aspx). They state that: "*At one time, oysters were so abundant in the Chesapeake Bay that their reefs posed a navigational hazard to ships sailing up the Bay. Now, because of disease, poor water quality, and decades of overharvest, the oyster population in the Bay is at about 1% of what it once was. Federal and state agencies, industry, academic institutions, and nonprofit groups have all been working hard to restore the native oyster population to levels that will once again provide the level of ecological and economic services that it once did.*"

As shown above (e.g. Dealteris 2004; Powers *et al.* 2007), shellfish aquaculture equipment can also serve as significant biogenic reefs which can increase the productivity of many invertebrates and fishes. Although artificial means of increasing oyster populations through aquaculture may not provide all functions of oyster reefs such as the 3-D habitat associated to natural reefs (Coen *et al.* 1999), oysters aquaculture can be considered of significant ecological benefit (Ulanowicz & Tuttle 1992). Aquaculture of the native oyster can also indirectly provide broodstock sanctuaries as bottom oyster populations are re-established. There are anecdotal reports of a number of bays where spawning and settlement of oysters have been restored, with the presence of water column oyster culture, where none had occurred for a few decades (C. Noris, personal communication).

5.6 Conclusion on Net Ecological Benefit Analysis

Bivalve culture, by its very nature, is an extractive activity where success is tied directly to environmental quality, natural supply of larvae and natural food availability. The FAO (2007) states that the "*Culture of molluscs is considered highly environmentally friendly as they do not require any inputs for growth and utilizes nutrients from the surrounding waters*". In addition to the value of the oysters themselves, the secondary productivity associated to the culture is also likely of significant value to fisheries resources (e.g. Powers *et al.* 2007).

We estimate that the natural population of oysters in N.B. estuaries at the turn of the 20th Century was approximately twenty times higher than current levels, including wild and aquaculture levels. Removal of endemic habitat created by oyster reefs has likely resulted in fragmentation, disturbance or elimination of ecosystem services, and net degradation of desirable estuarine functions. Newell (1988) suggested that the loss of oysters in Chesapeake Bay, due to disease and overfishing, contributed to undesirable ecosystem shifts in the food webs leading to a rise in the biomass of predators such as ctenophores and jellyfish. The author concluded that "an increase in the oyster population by management and aquaculture could significantly improve water quality by removing large quantities of particulate carbon".

There is mounting evidence that increasing the abundance of oysters is likely to restore some of the ecological services such as water filtration, benthic-pelagic coupling, and top-down control on phytoplankton once provided by natural stocks. These functions provide net benefits beyond the provision of fish habitat over an extended time-frame. Oysters in aquaculture structures are not considered different from oysters in nature. Thus, they can provide a number of ecological services, which can potentially increase the functional and structural sustainability of the ecosystem (Prins *et al.* 1997) and reduce the symptoms of ecosystem distress caused by eutrophication (Newell 1988, Jackson *et al.* 2001, Newell & Koch 2004).

Habitat restoration plans increasingly recognize the role of shellfish in improving water quality by assimilating and recycling large amounts of nutrients by feeding on plankton and thus aiding to mitigate the effects of anthropogenic eutrophication (Officer *et al.* 1982). Ferreira *et al.* (2007) discusses the economic potential for aquaculture operations as "nutrient sinks" to essentially remove the nutrient pollution from other industries and profit from this clean-up; similar to global emission trading mechanisms. In the U.S., in particular, where the loss of the American oyster has resulted in dramatic shifts in ecosystem equilibrium, there is consensus that restoration of oyster populations is critical in maintaining ecosystem health.

This Net Ecological Benefit Analysis allowed us to gain a greater perspective on elements which are not typically considered in an Ecological Risk Assessment. There remains a need to better understand how distinct habitat types, such as oyster reefs, interact within landscapes in order to better understand the contribution of aquaculture to supporting complex ecosystem linkages (Duffy 2006). The exercise of examining both positive and negative effects of shellfish aquaculture is informative, particularly in illustrating the challenge faced by managers in weighing the effects of certain activities. This is particularly true when the dynamics of this activity include non-linear relationships between multiple effects, both positive and negative, such as the ones associated with increasing shellfish abundance (Figure 8).

We conclude that, when properly managed, oyster aquaculture is likely to provide positive ecosystem services. This warrants further consideration as a key component in achieving healthy ecosystem objectives.

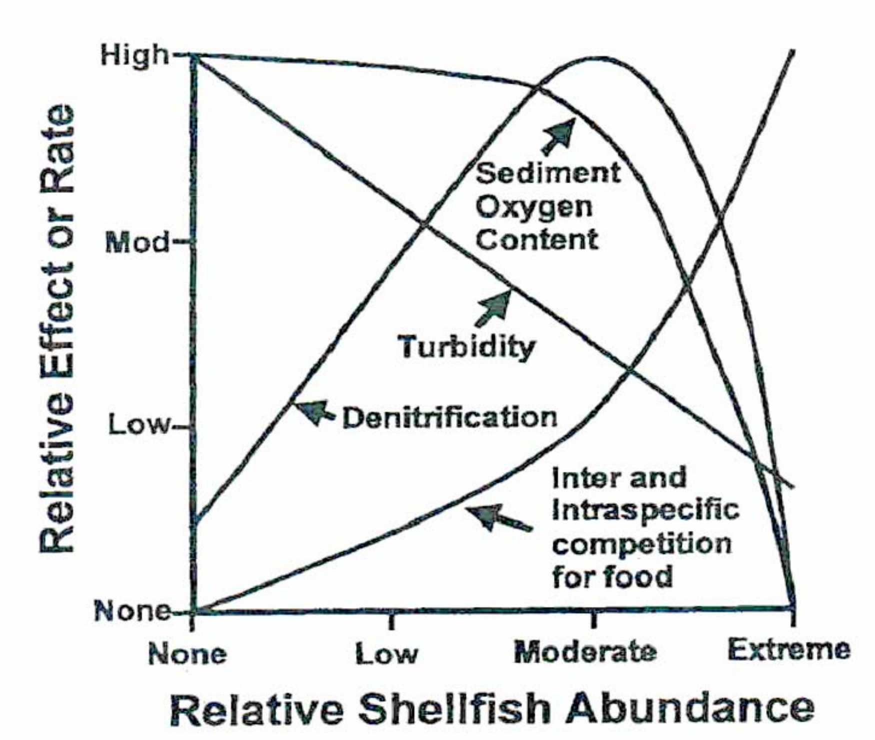

Figure 8 – Conceptual figure of relative effects associated to increased abundance of shellfish (from Newell 2004)

6 RISK MANAGEMENT ASSESSMENT

6.1 Identification of appropriate risk management options

The guiding principle for risk management is to achieve a reasonable degree of certainty that significant adverse effects can be avoided through a rationalised and feasible approach given the present knowledge limits, available options and resources. The HMP Risk Management Framework identifies a number of mechanisms to address low risk projects. Based on the framework and the perceived low risk associated with water column oyster aquaculture activity through the Ecological Risk Assessment, DFO considered the use of Operational Statements, letters of advice or Best Management Practices could have been acceptable options as operational tools to address the level of risk.

However, given the projected growth of the water column aquaculture industry, DFO Gulf Region favoured that EAs be managed by using the more rigorous Replacement Class Screening Report (RCSR) approach for this activity. This approach is built on the knowledge of the environmental effects of a given type of project while consolidating mitigation measures from governmental federal authorities involved in the process. A RCSR typically includes mitigation measures and Best Management Practices identical to those normally found in a site-by-site evaluation and letter of advice. This approach is also favoured because of the heightened public awareness and scrutiny surrounding aquaculture in general. The approach also implicitly requires that the authorities reflect on the activity in the context of their longer-term planning and bay-wide objectives as well as the acceptable levels of development that balance socio-economic and ecological sustainability.

As explained earlier, a replacement class screening consists of a single comprehensive report that defines the class of projects and describes the associated environmental effects, design standards and mitigation measures for projects assessed within the report. It includes a conclusion of significance of environmental effects for all projects assessed by the replacement class screening. This type of report presents a summary of the accumulated knowledge on the environmental effects of a given type of project and identifies measures that are known to reduce or eliminate the likelihood of these adverse environmental effects. A RCSR is also considered consistent with the more comprehensive Bay Management Framework (BMF), which constitutes a broader integrated planning and regulatory framework. In addition, a RCSR is a living document which includes provisions for revisions every five years, or whenever new information comes to light. Under a RCSR, yearly reporting of site review to the public registry is also required.

6.2 Risk Communication

Management of oyster aquaculture will require communication of the findings of this risk assessment. In N.B., like elsewhere in the world, the emergence of aquaculture as a relatively new and growing resource use can be perceived to be a disruption of the long-established *status-quo* between existing users (Burbridge *et al.* 2001,Shumway *et al.* 2003). The recent growth of aquaculture has occurred along coastlines where there is already a high concentration of other commercial, recreational and traditional resource users. This can provoke socio-economic concerns relating to aesthetics, property value and boating access, which is not unexpected, particularly in prime coastal real estate and recreational areas. In addition, the utilization of maritime space for aquaculture purposes raises potentially complex property and federal-provincial jurisdictional issues.

This risk assessment demonstrated that potential risks as they relate to fish and fish habitat have been identified and that the assessment of likelihood, consequences and probability of effects is based on reliable scientific evidence. The level of confidence in this approach is high, particularly in the context of a Bay Management Framework (BMF) where spatio-temporal interactions with ecological entities are reduced and/or avoided.

6.3 Risk monitoring, reporting and review

Research is being actively conducted by DFO, the Province of N.B., universities and the aquaculture industry itself. In August 2000, DFO launched its Program for Sustainable Aquaculture. The program reflects the federal government's commitment to increase scientific knowledge to support decision-making, strengthen measures to protect human health, and make the federal legislative and regulatory framework more responsive to public and industry needs. Specifically, the program allocates $75 million over five years with $15 million each year thereafter for: 1) environmental and biological science to improve the federal government's capacity to assess and mitigate aquaculture's potential impacts on aquatic ecosystems; 2) the Aquaculture Collaborative Research and Development Program, under which DFO partners with industry by jointly funding R&D projects to enhance sector innovation and productivity; 3) strengthening of the Canadian Shellfish Sanitation Program; 4) enhancement of the application of DFO's legislation, regulations and policies that govern aquaculture, particularly as they relate to habitat management and navigation.

Additionally, monitoring programs are ongoing in order to collect baseline data. For example, the Shellfish Monitoring Network has standardised cages housing mussels or oysters in multiple bays in the Maritime Provinces to provide a baseline of shellfish productivity. Also the Community Aquatic Monitoring Program's (CAMP) is being conducted in 26 sites in the Maritimes. CAMP is being used to build working relationships between DFO and community environmental groups, academia and other interested parties as well as to collect information on fish and invertebrate communities, water quality (e.g. temperature, pH, nutrients, etc.) and aquatic vegetation with the collaboration of watershed groups in several bays.

The development of the bivalve aquaculture industry is being closely supervised in N.B. The New Brunswick Shellfish Aquaculture Environmental Coordination Committee (NBSAECC) provides a forum for inter-agency communication which tracks the continuously evolving scientific and technical knowledge related to the activities of this sector and can recommend changes in shellfish aquaculture management practices when needed. Representatives of DFO, the Province of N.B., Transport Canada, Environment Canada as well as the New Brunswick Professional Shellfish Growers Association (NBPSGA) sit on this committee.

Yearly, through the *Canada-N.B. MOU for Aquaculture Development*, the NBSAECC meets to review the data resulting from field surveys and research conducted by academics, federal and provincial agencies. If significant changes occur in the risk posed by the husbandry methods (e.g. appreciable changes in intensity or techniques), the environmental conditions (e.g. water quality), or in the state of knowledge concerning water column oyster aquaculture, they are required to report updated assessments to senior managers of their respective agencies. The Canada-N.B. Aquaculture Management Committee can thereafter make decisions to address concerns.

Additionally, the BMF developed with the Province of N.B. is an example of a living tool and is based on the premises of Adaptive Management to ensure the sustainable development of the shellfish aquaculture sector. A management team has been established to regularly review the outcome of the overall planning and regulatory framework to ensure it is regularly adapted. The team will continue to evaluate the effectiveness of the BMF in regards to integrated sustainable aquaculture development, based on sound planning and management.

7 CONCLUSIONS

The Habitat Management Program's Risk Management Framework implicitly recognises that all activities entail some risks which must be weighed in terms of the *scale of negative effect* and the *sensitivity of fish and fish habitat* using the Risk Assessment Matrix (Figure 3). The Ecological Risk Assessment characterizes many of the risks and assesses their significance in the context of the scientific literature and ecosystem dynamics; in summary we conclude that:

- The overall *scale of potential negative effects* of water column oyster aquaculture in N.B. is low. In general the *sensitivity of fish and fish habitat* is low, eelgrass which is being affected by a number of anthropogenic impacts is considered moderately sensitive. For that reason oyster aquaculture works in N.B correspond to a low-risk activity on the HMP Risk Assessment Matrix;

- Given the low densities observed in water column oyster aquaculture in N.B., which differ greatly from other regions in the world, for an activity where "*most effects of bivalve aquaculture seem to be related to the scale (intensity and extent) of aquaculture rather than the type of infrastructure*" (DFO 2006), the potential for significant residual effects after mitigation is low;

- Thus the activity is considered unlikely to significantly harm the productive capacity or the ecological integrity of fish habitat. The risks associated with water column oyster aquaculture can be managed with adequate planning and mitigation measures through an adaptive management approach.

The development of this risk assessment has lead to the evaluation of a number of potential management tools available within DFO's regulatory mandate. Given the conclusion on the level of risk, the use of Operational Statements, Best Management Practices, etc is considered adequate. Because of the heightened public awareness and scrutiny surrounding aquaculture in general, the use of a RCSR is considered a prudent and appropriate operational tool for integrating several regulatory and expert advices of federal departments to manage the level of risk to fish and fish habitat posed by the oyster aquaculture industry.

Although the risk analysis framework generally focuses on negative effects and does not presently integrate the Net Ecological Benefit Analysis into the decision-making process, we found the exercise to be informative with regards to evaluating the complexities in ecosystem dynamics and in qualifying the overall effects of this activity. Accordingly, we believe that shellfish aquaculture, when managed effectively, can provide many ecosystem benefits and can contribute to the general environmental health of N.B. estuaries. The Net Ecological Benefit Analysis also served to illustrate how our current view of temperate estuaries in our region is that of an altered state (i.e. depleted oyster reefs) in comparison with the reference state which was dominated by extensive bivalve meta-populations. This conclusion supports the general approach taken by the HPSD of recommending the development of oyster reefs as compensation projects for habitat losses. These types of considerations will likely become increasingly important as governments continue to work towards planning and implementing a more formal ecosystem approach to managing coastal activities based on regional objectives of sustainable development.

8 ACKNOWLEDGEMENTS

Firstly, we thank all the participants of the *National Science Workshop: Assessing Habitat Risks Associated With Bivalve Aquaculture in the Marine Environment* National Assessment Process. This workshop and all the related documents that were produced largely inspired this risk assessment.

We are also grateful to Chad Ziai and Roland Cormier for their assistance and expertise in the Habitat Management Program and the Risk Assessment Framework.

Special thanks go to Sylvio Doiron, Abel Noël, Dr. Luc Comeau, Claire Carver and Dr. André Mallet who all contributed to this report with their vast knowledge on oyster aquaculture in New Brunswick.

We would especially like to thank Dr. Roger Newell and Dr. Christopher Pearce for their peer-review as well as their constructive comments which were extremely helpful in finalizing this risk assessment.

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Aquaculture 410–411 (2013) 57–65

Contents lists available at ScienceDirect

Aquaculture

journal homepage: www.elsevier.com/locate/aqua-online

Suspended versus bottom oyster culture in eastern Canada: Comparing stocking densities and clearance rates

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ARTICLE INFO

Article history:
Received 22 March 2013
Received in revised form 17 June 2013
Accepted 19 June 2013
Available online 28 June 2013

Keywords:
Aquaculture
Oyster
Clearance
Density

ABSTRACT

The objectives of this study were to compare the stocking density of suspended versus bottom oyster (*Crassostrea virginica*) culture in Atlantic Canada and to estimate the capacity of these oysters to clear particles from the water column. Surveys of multiple leases indicated that stocking densities for floating bag and floating cage culture techniques were on average 0.3 ± 0.1 and 0.5 ± 0.1 kg oysters m^{-2}, respectively. Bottom culture density was estimated at 1.0 ± 0.1 kg oysters m^{-2}, whereas natural reef density was assessed at 2.2 ± 1.1 kg oysters m^{-2}. In terms of grazing potential, suspended oysters had significantly lower gill areas per unit dry tissue weight than bottom oysters. This result was consistent with power functions relating clearance rate (CR, $l\ h^{-1}$) to dry tissue weight (DTW, g). CR increases relative to DTW were significantly lower in the suspended oyster category than in the bottom oyster category, as indicated by the exponent in the relationships $CR = 6.35 \pm 0.59 \times DTW^{0.78 \pm 0.08}$ (bottom) and $CR = 4.34 \pm 0.32 \times DTW^{0.41 \pm 0.08}$ (suspended). Based on this information it was calculated that CR per unit area (CR_{Area}) in the most heavily exploited leases was 66.5 ± 8.5 (floating bags), 86.5 ± 8.6 (floating cages), and 197.3 ± 144.4 (bottom culture) $l\ h^{-1}\ m^{-2}$. The CR_{Area} for suspended techniques was on average 10 to 14 times lower than the CR_{Area} for healthy oyster reefs. A bay-scale assessment of an intensive culture site led to the conclusion that cultivated oysters do not exert a dominant top-down control on phytoplankton abundance.

1. Introduction

The Eastern oyster, *Crassostrea virginica* (Gmelin 1791), has a remarkable latitudinal distribution range along the Northwest Atlantic seaboard. Native populations are found in the Gulf of Mexico (27°N) and northward into the Gulf of St. Lawrence (48°N), Canada (Carriker and Gaffney, 1996). In the two Canadian provinces of Prince Edward Island (PEI) and New Brunswick (NB), oyster farming was first started in 1865, when seed collected from natural reefs were transplanted to leased bottom areas for the purpose of rearing oysters to commercial size (Lavoie, 1995; Mathieson, 1912). However, since the early 1990s, the traditional approach of relaying seed to bottom culture areas is being progressively replaced by suspended culture. Novel suspension techniques are being developed using various types of holding compartments. The most popular types are UV-resistant polymer mesh bags often referred to as Vexar™ bags. These bags can be equipped with individual floats (Fig. 1), or inserted into wire-mesh cage structures equipped with large floats (Fig. 2). Floating bags and cages are attached to longlines deployed in the subtidal zone where they can be flipped (180°) and temporarily exposed to the air to desiccate fouling organisms (Mallet et al., 2009). Prior to the onset of winter and the formation of a thick (~1 m) ice cover, entire longlines of bags or cages are lowered onto the bottom either by removing the floats, or flooding the flotation compartments.

From a farming perspective, there are several advantages to suspending oyster stocks in the upper water column. This strategy protects stocks from benthic predators and facilitates product grading and harvesting procedures. Also, the relatively warm and elevated food flux environment in the upper water column (Comeau et al., 2010) enhances growth (Bataller et al., 1999) and shortens the production cycle (Doiron, 2008). Oysters grown in suspension generally reach market-size within 3 to 4 years, which is much faster than the 5 to 8 years normally required when grown on the substrate. Finally, oysters grown in suspension are morphologically similar to those growing at low densities on firm bottoms. They have a tendency to develop round shells ornamented with radial ridges and foliated processes (Galtsoff, 1964). By contrast, oysters grown on soft, muddy bottoms tend to develop elongated and sparsely ornamented shells (Fig. 3).

The first objective of this study was to gain a better empirical understanding of the stocking density of suspended oyster culture in Atlantic Canada. Presently, information is lacking as to whether suspended leases are exploited to their full capacity. Based on their dimensions and mooring requirements, the floating bag technique allows a maximum deployment of 2000 bags ha^{-1} (Doiron, 2008). Similarly, floating cage mooring guidelines dictate a maximum stocking density of approximately 1500 bags ha^{-1}. However, it remains unclear how

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http://dx.doi.org/10.1016/j.aquaculture.2013.06.017

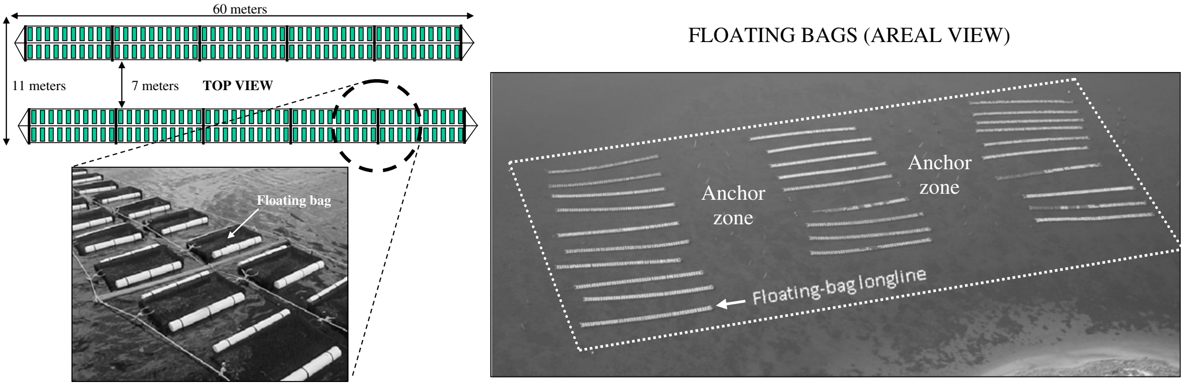

Fig. 1. Left: individual floating bags (45.7 cm × 88.9 cm × 7.6 cm) holding cultivated oysters and attached to a longline system (modified from Doiron, 2008). Right: aerial view of an entire lease (24,380 m^2) containing multiple longlines in the subtidal zone (modified from Comeau et al., 2006).

these suspended culture densities compare with those found in bottom culture beds and natural reefs. Perhaps an impeding difficulty is that information on bottom oysters in eastern Canada is restricted to a grey literature that for the most part is difficult to trace.

The second objective of this study was to gauge the ability of cultivated oysters to clear particles from the water column. It could be postulated that suspended oysters have high clearance rates given that they grow relatively fast. Over the past century, pumping rates and clearance rates have been sporadically reported for wild *C. virginica* collected in the intertidal or subtidal zone (Galtsoff, 1926; Loosanoff, 1958; Palmer, 1980; Riisgard, 1988), with 10 l h^{-1} g^{-1} dry tissue weight documented as a maximum clearance rate (Eastern Oyster Biological Review Team, 2007). To my knowledge, the clearance rates of suspension-grown *C. virginica* were investigated in a single laboratory study which reported a maximum rate of 4 l h^{-1} g^{-1} dry tissue weight (Pernet et al., 2007).

2. Methods

2.1. Suspended oyster survey

A total of 133 suspended leases were surveyed across 20 embayments in NB and PEI in 2011–2012 (Fig. 4). All leases were surveyed by boat in early autumn, prior to the lowering of the gear onto the bottom to avoid winter ice. Floating bags, including those contained in floating cages, were counted in each lease. Bag content in terms of number and size of oysters was estimated based on standard husbandry practices (Doiron, 2008). Suspended leases typically hold four year classes distributed according to the proportions given in Table 1. Using this information it was calculated that a normalized bag contains 332 oysters, which weigh a total of 6.1 kg (see Table 1 caption for details). Lease-scale oyster density (OD) was calculated as follows:

$$\mathrm{OD} = \frac{N_{\mathrm{Bag}} \times 6.1}{\mathrm{Area}}$$

where OD represents oyster biomass (kg) m^{-2}, N_{Bag} is the number of bags counted within the lease, Area is the lease area (m^2), and 6.1 is the normalized oyster weight (kg) in each bag.

2.2. Bottom oyster survey

A total of 10 grey literature reports provided a detailed description of natural oyster reefs and leased bottom areas in eastern NB (Table 2). These surveys were conducted between 1974 and 2001, and the number of sites investigated ranged from 1 to 11. In all cases, live oysters within 0.12 to 1-m^2 quadrats were removed by hand, counted, and weighed (whole weight). When whole weight

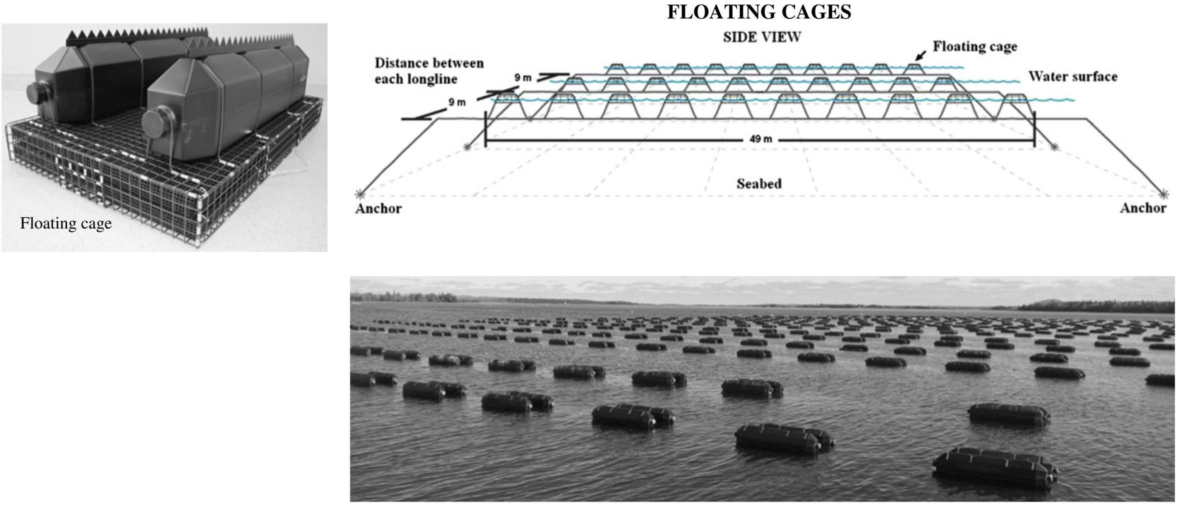

Fig. 2. Left: wire-mesh OysterGro™ floating cage (147.3 cm × 91.4 cm × 15.2 cm) housing 6 individual Vexar bags (45.7 cm × 88.9 cm × 7.6 cm). Dented triangular structures on top of buoys deter birds from roosting on the structure. Right (top): sketch of recommended mooring layout for floating cages (Bouctouche Bay Industries Ltd., Bouctouche, NB, Canada). Right (bottom): photograph of floating cages in Foxley River, PEI.

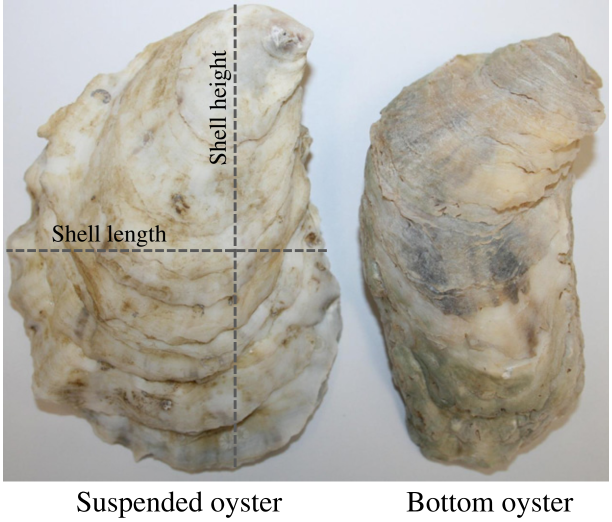

Fig. 3. Typical shape of suspended and bottom-grown oysters in eastern Canada. In this example, shell height and shell length, defined according to Galtsoff, 1964, are 88 × 64 mm (suspended) and 84 × 39 mm (bottom).

information was lacking in the report, it was calculated using the following allometry relationship (Landry et al., 2001):

$$\textit{whole weight in g} = 2.90\ \textit{Log}_{10}(\text{shell height in mm}) - 3.84\ \left(r^2 = 0.97,\ n = 152\right)$$

Since no report was available for PEI, 26 quadrats were sampled across four bottom leases in Foxley River in July 2012. At each station, a 0.5-m^2 quadrat was thrown over the side of the boat and a SCUBA diver then collected all the live oysters within the quadrat. Oysters were counted and measured for whole weight, shell height, and dry tissue weight. Mean density in terms of oyster whole weight (kg m^{-2}) was calculated for each bottom lease.

Table 1
Typical year class distribution in a suspended culture lease (Doiron, 2008). A normalized bag contains 332 oysters[a] and weighs 6.1 kg[b].

Year class	Shell height (mm)	Weight (g)	DTW (g)	Number per bag	Percent bags in lease
Y1	<31	1.77	0.04	1000	8
Y2	31–50	8.80	0.14	500	17
Y3	51–65	22.80	0.36	250	33
Y4	>65	39.10	1.57	200	42

[a] (1000 × 0.08) + (500 × 0.17) + (250 × 0.33) + (200 × 0.42).
[b] ((1000 × 0.08 × 1.77) + (500 × 0.17 × 8.8) + (250 × 0.33 × 22.8) + (200 × 0.42 × 39.1))/1000.

2.3. Clearance rates

Clearance rate (CR) is defined as the volume of water cleared of suspended particles per unit time. In this study, maximum CR was measured as part of a controlled comparison of bottom and suspended oysters fed a natural diet. On 28 September 2012, oysters of varying sizes were collected in Foxley River PEI and brought to a field laboratory in Georgetown PEI where they were held in a large tank (250 L) continuously supplied with natural seawater (temperature ~16 °C). After a one-week acclimation period, 10 oysters (5 bottom and 5 suspended) were transferred to individual acrylic chambers supplied with the same seawater as the holding tank. The chamber volume selected (190, 670 or 1100 ml) was dependent on the size of the oyster. Two additional chambers containing shells only served as controls to measure gravitational settling of particles. Particle mixing was promoted by fine bubble aeration, introduced in a manner that minimized the resuspension of feces. Each chamber was equipped with a fluorometer (CYCLOPS-7® submersible sensor, Turner Designs, Sunnyvale, CA) connected to a data acquisition controller with software (Microlink 751, Windmill Software Ltd, Manchester, UK) that provided a quasi-real time (5 s delay) graphical display of fluorescence. Following a 1-hour adaptation period, water flow was halted and the decline in fluorescence over time was monitored on the computer screen. Any oysters that expulsed chlorophyll material into their chamber, creating major spikes in the fluorescence readings, were excluded from the experiment. Only chambers that showed a continuous exponential decrease in fluorescence over time were included in the final analysis. This standardization approach minimized the potential underestimation of CR. Particle depletion rates within the chambers were measured by counting suspended particles at the start of the static incubation and approximately 10 min later. Water samples (10 mL) were extracted from the chambers and aliquots (100 µL) were processed using a Beckman Coulter Counter Z1^{TM} fitted with a 100-µm aperture tube. The instrument was set to measure particles in the size range of 5–19 µm, which

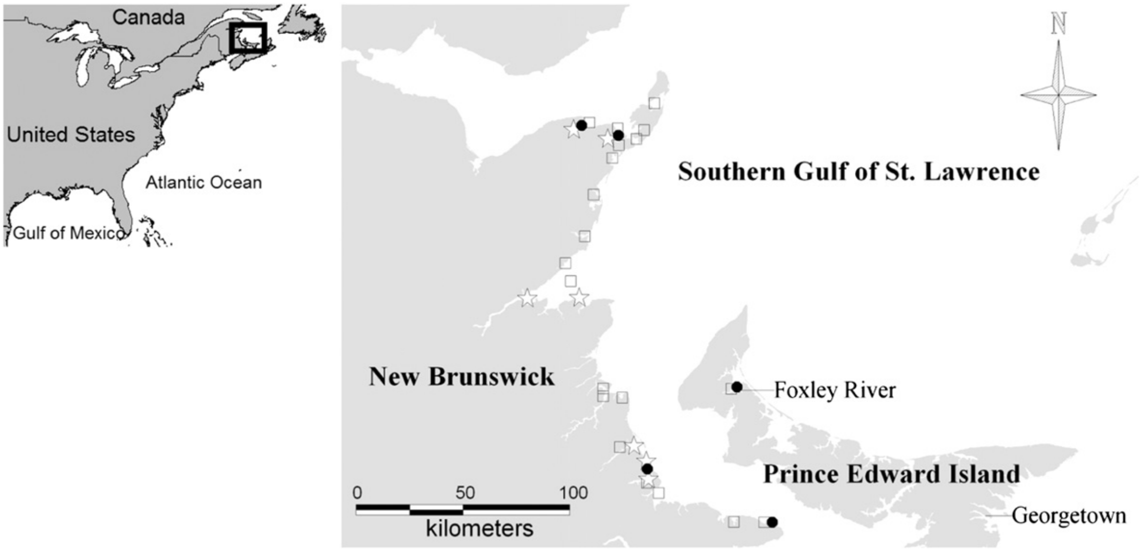

Fig. 4. Map of study area showing sampling locations for suspended culture (□), bottom culture (•) and natural reefs (☆).

Table 2
Summary of the dataset that was built for the meta-analysis. n refers to the number of oyster leases or oyster reefs sampled.

Lease type	Survey source	Survey year	n	Survey sites
Suspended	This study	2011	111	19 bays, NB
	This study	2012	32	Foxley River, PEI
Bottom	Ferguson (1983)	1983	11	St. Simon, NB
	Ferguson (1985)	1984	5	Caraquet, NB
	McIver and Woo (1975)	1972	6	Bouctouche, NB
	Doiron (1992)	1990	1	Spence Cove, NB
	This study	2012	4	Foxley River, PEI
Natural reefs	Milewski and Chapman (2002)	2001	10	4 bays, NB
	SEnPAq (1990)	1990	2	Miramichi, NB
	Landry et al. (2001)	1999	1	Caraquet, NB
	Sephton and Bryan (1988)	1987	1	Caraquet, NB
	Lavoie and Robert (1981)	1979	1	Caraquet, NB
	Lavoie (1977)	1974	1	Caraquet, NB

are known to be completely retained by oysters (Riisgard, 1988; Ward and Shumway, 2004). CR was calculated according to the formula:

$$\mathrm{CR} = {}^{V}\!/_{t} \times \ln {}^{C_0}\!/_{C_t}$$

where V is the volume of the chamber, t is time elapsed between measurements, and C_o and C_t are particle concentrations at times 0 and t, respectively (Riisgard, 1988, 2001). At the end of the incubation period, oysters were removed from the chambers and replaced with new specimens taken from the holding tank. These trials were repeated until CR was successfully measured on 39 bottom oysters and 29 suspended oysters. Shell height, whole weight, and dry tissue weight (DTW) were determined for each individual.

Given that shell height and whole weight were poor predictors of CR, power equations describing CR as a function of DTW were established for bottom and suspended oysters. These equations were used to calculate a CR per unit surface area ($\mathrm{CR_{Area}}$) for each of the culture categories under investigation. For suspended culture, CR was first scaled up to a normalized bag based on the number of oysters, the year-class proportions and the DTW values provided in Table 1. $\mathrm{CR_{Area}}$ was then calculated as follows:

$$\mathrm{CR_{Area}} = \frac{\mathrm{CR_{Bag}} \times \mathrm{N_{Bag}}}{\mathrm{Area}} \times 0.686$$

where $\mathrm{CR_{Area}}$ represents CR per unit leased area ($\mathrm{l\ h^{-1}\ m^{-2}}$), $\mathrm{CR_{Bag}}$ is the normalized bag CR ($\mathrm{l\ h^{-1}}$), $\mathrm{N_{Bag}}$ is the number of bags counted within the lease, Area is the lease area ($\mathrm{m^2}$), and 0.686 represents the proportion of time (68.6%) oysters have their valves open when feeding on natural seston (Comeau et al., 2012).

Similar calculations were made for assessing CR in bottom culture ($\mathrm{l\ h^{-1}\ m^{-2}}$) and natural reefs ($\mathrm{l\ h^{-1}\ m^{-2}}$). CR for each individual collected in the survey quadrats was calculated based on the CR–DTW relationship developed for bottom oysters. Where DTW was lacking from survey datasets, a power equation was used to convert reported shell heights into DTW. This relationship was based on a sample of bottom oysters (DTW 0.4–3.7 g) collected in Foxley River in September 2012.

$$\mathrm{DTW} = 0.0002 \times \text{shell height}^{1.9192} \quad \left(r^2 = 0.75,\ n = 94\right)$$

The sum of individual CR for each quadrat was normalized to a single square meter and then multiplied by 0.686 to correct for the proportion of time oysters are typically feeding (Comeau et al., 2012). Results from individual sampling sites (quadrats) were average to produce lease or reef-scale estimates of CR ($\mathrm{l\ h^{-1}\ m^{-2}}$).

2.4. Gill area

In September 2012, bottom and suspended oysters were collected in Foxley River to investigate whether morphological differences existed between the two categories, specifically in regards to the size of their gills. Gills were excised from 24 bottom and 27 suspended oysters. Gill area ($\mathrm{Gill_A}$) was assessed by digital image analysis following Honkoop et al. (2003). Once this analysis was completed, gills and other soft tissues were pooled to determine individual DTW. Shell height was also measured.

2.5. Statistics

All statistical analyses were conducted using SPSS v. 20 software (IBM SPSS, Armonk, NY, USA). Regression analysis was used to explore relationships between lease area (ha) and farming activity within leases, namely oyster biomass (tons) and oyster stocking density ($\mathrm{kg\ m^{-2}}$). Serial independence of the error terms was graphically assessed and further tested using the Durbin–Watson test; residuals were screened for normality using expected normal probability plots and the Kolmogorov–Smirnov test. Residuals were both graphically and quantitatively (Levene test) assessed for homogeneity of variances. Data were sqrt-transformed or log-transformed where heteroscedasticity was detected. When data transformations failed to stabilize the variance, weighted-least square regression analysis was applied, in which case, weights were estimated by examining the relationship of the variance of the dependent variable to various powers of values of the independent variable.

Homogeneity of regression slopes was tested on log-transformed data using the SPSS GLM procedure. The oyster category (suspended, bottom) was set as a fixed factor and the independent variable was identified as a covariate. Where the homogeneity of regressions was not rejected, an ANCOVA was performed to test whether oyster origin had a significant effect on the dependent variable.

A mixed model analysis of variance (procedure GLM in SPSS) was developed to test the effect of oyster category (C) on oyster stocking densities (OD). C included four categories: natural reefs, bottom culture, floating bag culture, and floating cage culture. In the model, C was declared a main effect (C [$i = 1$ to 4]) and the data source (S) was set as a random effect (S [$j = 1$ to 10]). OD was rank-transformed to stabilize the variance (Levene's test, $P > 0.05$).

$$\mathrm{OD_{ij}} = \mu + \mathrm{C_i} + \mathrm{S}_j(\mathrm{C}_i) + \varepsilon_{\mathrm{ij}}$$

A second mixed model was developed to test the effect of C on the $\mathrm{CR_{Area}}$. This analysis was restricted to leases that had the most oysters per unit area, and therefore that were exploited at, or near their full capacity. The same logic was applied to natural reefs by selecting the most densely populated examples. In keeping with this rationale, stocking density values above the 75th percentile for each category were selected for analysis. In the model, C was declared a main effect (C [$i = 1$ to 4]) and the data source (S) was set as a random effect (S [$j = 1$ to 10]). CR_{Area} was rank-transformed to stabilize the variance (Levene's test, $P > 0.05$).

$$CR_{Area\ ij} = \mu + C_i + S_j(C_i) + \varepsilon_{ij}$$

When the main effect, C, was significant, Tukey's HSD post hoc tests were performed to determine homogeneous groups. In this paper, all measures of variability reported along with the mean values represent 1 standard error of the mean (mean ± 1 SEM).

3. Results

3.1. Oyster densities

Of the 133 suspended leases surveyed, 123 were classified as being active, i.e., containing oysters that were suspended in the water column by some means. While a small proportion of these oysters were suspended using strings, tables, racks or other means, the bulk (95.4%) of the surveyed stock was contained in floating bags or floating cages.

Further analyses were conducted on leases containing exclusively floating bags ($n = 48$ leases) or floating cages ($n = 39$ leases). It was first examined whether allocated lease area, a metric which is readily available from licensing departments, is a good indicator of the farming activity level within the lease, either in terms of total biomass or stocking density. No significant correlations were found between lease area and farming activity metrics for leases containing floating bags. However, for leases populated with floating cages, lease area was a weak but significant predictor of total biomass (Fig. 5a, $r^2 = 0.28$, $P < 0.01$, weighted least squares) and stocking density (Fig. 5b, $r^2 = 0.38$, $P < 0.01$, ordinary least squares). In general, smaller leases tended to be more densely stocked than larger leases.

Stocking densities for floating bag and floating cage techniques were respectively 0.3 ± 0.1 and 0.5 ± 0.1 kg oysters m^{-2} (Fig. 6); the difference in stocking density between gear types was significant. It is also noteworthy that both average densities were below prescribed mooring deployment guidelines for suspended culture. In relative terms, floating bag and floating cage densities were 77.2 ± 3.7% and 47.6 ± 6.1% below the recommended level, respectively.

Based on the NB grey literature and the Foxley River data, bottom culture density was estimated at 1.0 ± 0.1 kg oysters m^{-2}. This estimate is significantly higher than those for suspended techniques. Bottom culture densities were statistically similar to those found in natural reefs, which averaged 2.2 ± 1.1 kg oysters m^{-2}. The elevated variance in the latter category is mainly attributable to two highly aggregated reefs (8.4 and 14.3 kg oysters m^{-2}).

3.2. Clearance rate (per unit body weight)

Power functions relating CR (l h^{-1}) to DTW (g) were calculated for bottom ($CR = 6.35 \pm 0.59 \times DTW^{0.78 \pm 0.08}$, $r = 0.85$, $P < 0.001$) and suspended ($CR = 4.34 \pm 0.32 \times DTW^{0.41 \pm 0.08}$, $r = 0.71$, $P < 0.001$) oysters (Fig. 7a). The exponent describes how fast CR increases relative to body weight. The hypothesis of equal slopes (exponent) between the two oyster categories was rejected ($P = 0.002$). CR increases relative to body size were significantly lower in the suspended category than in the bottom category, starting at a dry tissue weight of approximately 0.25 g. The equations predict that a 1 g DTW bottom oyster has a CR of 6.3 l h^{-1} whereas a suspended oyster of comparable DTW has a CR of 4.3 l h^{-1}.

CR results are consistent with gill measurements taken on a sample of large oysters (Fig. 7b). The relationship between gill area (cm^2) and DTW (g) was best described as $Gill_A = 12.27 \times DTW^{0.59 \pm 0.06}$ ($r = 0.90$, $P < 0.001$) and $Gill_A = 9.93 \times DTW^{0.61 \pm 0.05}$ ($r = 0.92$,

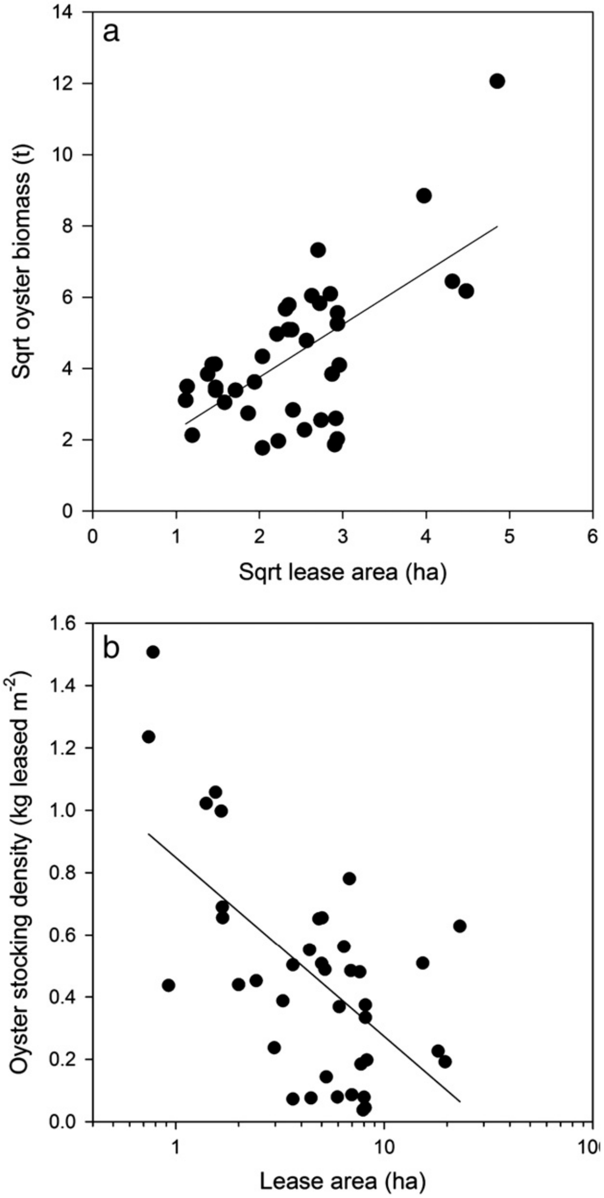

Fig. 5. Oyster biomass (a) and oyster stocking density (b) as a function of floating-cage lease area.

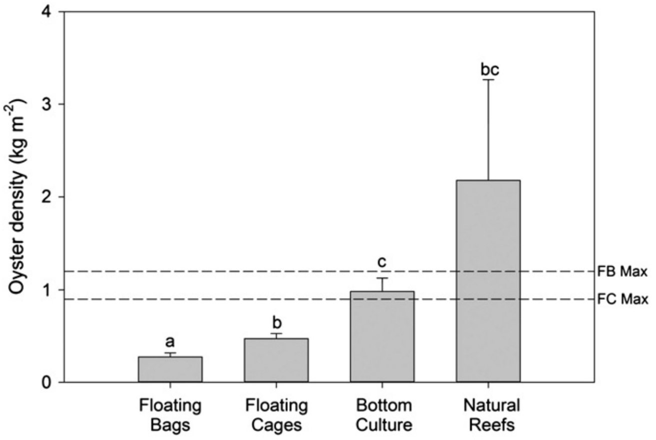

Fig. 6. Oyster density (mean ± SEM) for different culture techniques and natural reefs. Reference lines show prescribed floating bag (FB) and floating cage (FC) densities based on gear mooring guidelines from manufacturers. Tukey's HSD test, $P < 0.05$.

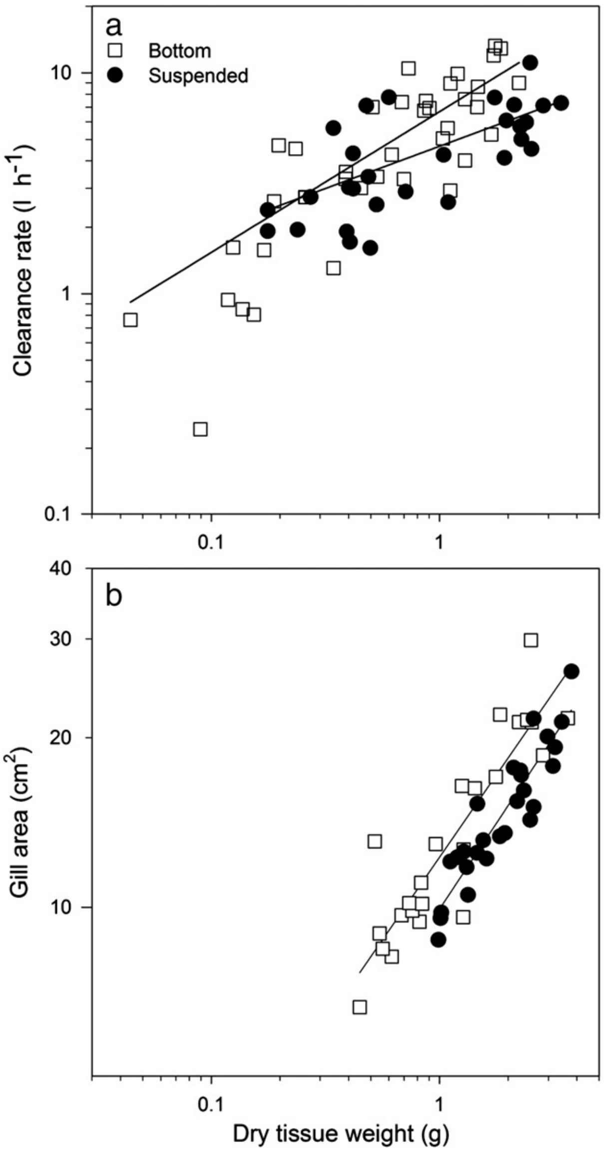

Fig. 7. Clearance rate (a) and gill area (b) in relation to dry tissue weight of bottom and suspended oysters. Note that gill area was measured on samples containing only large oysters.

$P < 0.001$) for bottom and suspended oysters, respectively. The hypothesis of equal slopes was not rejected ($P = 0.77$). However, suspended and bottom oysters had significantly different gill areas per unit body weight (ANCOVA, $P < 0.001$). Gill area standardized to an oyster of 1 g DTW was 12.3 cm^2 for the bottom category and 9.9 cm^2 for the suspended category.

Suspended oysters were in good physiological condition, i.e., they had elevated DTW values. This observation became evident after plotting DTW against shell height, a size indicator commonly used by field observers. The plot shows large suspended oysters (>60 mm) having higher DTW values than bottom oysters of comparable shell height (Fig. 8). The hypothesis of equal slopes (exponent) between the two oyster categories was rejected ($P < 0.001$): DTW increases relative to shell height were significantly higher in the suspended category than in the bottom category.

3.3. Clearance rate (per unit area)

Fig. 9 shows CR$_{\text{Area}}$ for the most densely populated (>75th percentile) leases and natural reefs contained in the dataset. CR$_{\text{Area}}$ for the floating-bag leases was 66.5 ± 8.5 l h^{-1} m^{-2}, or approximately 42% below the CR$_{\text{Area}}$ expected for this technique assuming full exploitation (based on mooring guidelines). CR$_{\text{Area}}$ for floating-cage leases was 86.5 ± 8.6 l h^{-1} m^{-2}, consistent with a full exploitation of this technique (84.5 l h^{-1} m^{-2}). The CR$_{\text{Area}}$ calculated for suspended techniques was on average 10 to 14 times lower than the CR$_{\text{Area}}$ for natural oyster reefs.

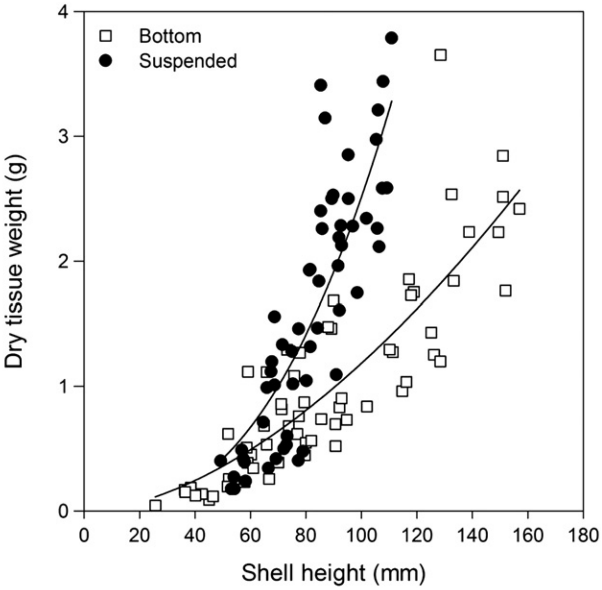

Fig. 8. Measured dry tissue weight in relation to shell height of bottom and suspended oysters.

The range of possible CR$_{\text{Area}}$ values for suspended culture was calculated by assuming all bags contain one of four year classes (Table 3). It was found that CR$_{\text{Area}}$ ranges from 41.3 ± 5.2 l h^{-1} m^{-2} where all floating bags contain small (< 31 mm) oysters, to 108.5 ± 10.8 l h^{-1} m^{-2} where all floating cages contain large (>65 mm) oysters.

4. Discussion

4.1. Oyster stocking densities

Oyster tables in France's Normandy area support approximately 6 kg oysters m^{-2} (*Crassostrea gigas*), assuming a restrained deployment of

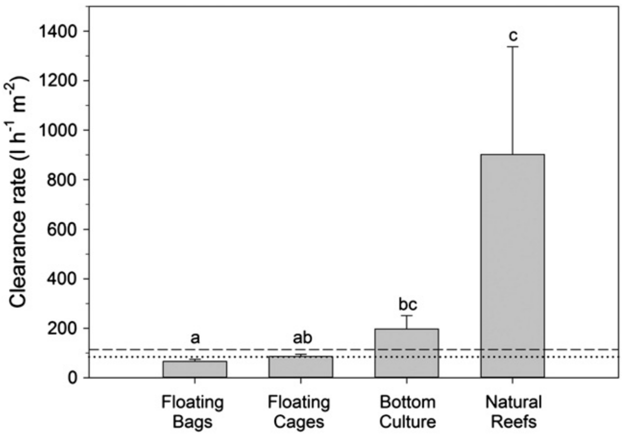

Fig. 9. Clearance rate (mean ± SEM) for the most densely populated (>75th percentile) leases and natural reefs. Dotted lines represent theoretical maximum for the floating bag (—) and floating cage (····) techniques based on mooring guidelines from gear manufacturers. Tukey's HSD test, $P < 0.05$.

Table 3
CR_{Area} (mean ± SEM) for the most densely populated (>75th percentile) suspended leases. Results show scenarios where all bags in leases contain a single year class (Y1, Y2, Y3 or Y4), and also the average scenario (Y1–Y4) based on year class proportions indicated in Table 1.

Year class	Shell height (mm)	Number oysters per bag	CR_{Bag} ($l\ h^{-1}$)	Floating bag CR_{Area} ($l\ h^{-1}\ m^{-2}$)	Floating cage CR_{Area} ($l\ h^{-1}\ m^{-2}$)
Y1	<31	1000	515.6	41.3 ± 5.2	53.7 ± 5.3
Y2	31–50	500	685.3	54.8 ± 7.0	71.3 ± 7.1
Y3	51–65	250	714.2	57.2 ± 7.3	74.3 ± 7.4
Y4	>65	200	1042.9	83.5 ± 10.6	108.5 ± 10.8
Y1–Y4	various	332	831.4	66.5 ± 8.5	86.5 ± 8.6

5000 bags ha^{-1} × 12 kg oysters bag^{-1} (Kopp et al., 2001). Here it is reported that densities for floating bag and floating cage culture techniques in eastern Canada were on average ≤ 0.5 kg oysters m^{-2}. Interestingly, the recent transition to suspended culture resulted in an actual reduction in stocking density compared to traditional bottom culture operations. Densities of 0.3 ± 0.1 and 0.5 ± 0.1 kg oysters m^{-2} were recorded for floating bags and floating cages, compared to 1.0 ± 0.1 kg oysters m^{-2} for the more traditional bottom culture. Suspended culture densities were generally well below technical guidelines prescribed by gear developers, suggesting that leases were underexploited and that the industry was still undergoing a developmental phase. Only a fully exploited suspended lease, one containing floating gear moored according to guidelines throughout its entire area, would compare with a bottom lease in terms of stocking density.

The reported densities for suspended culture are comparable to natural populations identified as being in a precarious state, which are presently being targeted for repletion and restoration. In Chesapeake Bay, for example, median values for live oyster abundance were 0.3 (Southworth et al., 2010) and 0.7 (Mann et al., 2009) kg m^{-2} from 1993 to 2009. Maximal abundance was approximately 8 kg m^{-2} for the 30 reefs surveyed in these two papers and 14.3 kg per m^2 for the dozen reefs reported in the present study. Unfortunately, absolute oyster densities prior to the degradation of natural reefs by destructive fishing practices, siltation, diseases and other habitat disturbances are poorly quantified. At the turn of the 20th century, when a number of oyster reefs were already considered depleted over the continental margin of North America (Kirby, 2004; Morse, 1971), Bastien-Daigle et al. (2007) estimated that there would have been a standing stock in the order of 35,912 t in NB. In 2005, the standing stock of cultured oysters in this province was evaluated at 1,249 t (Comeau et al., 2006), or approximately 3.5% of the historical standing stock of oyster reefs.

4.2. Clearance rate (per unit body weight)

Estimates of CR for *C. virginica* were summarized in Grizzle et al. (2008). CR standardized to an oyster of 1 g DTW was reportedly 6.79 L h^{-1} (Riisgard, 1988), 6.40 l h^{-1} (Newell and Koch, 2004), and 7.46 to 9.62 l h^{-1} (Newell et al., 2005). These values were derived from wild oysters primarily feeding on laboratory diets under optimal conditions (20–29 °C). CR for bottom-cultivated oysters acclimated to 16 °C and grazing on natural seston were consistent with this literature. A 1 g DTW bottom-cultivated oyster had a CR of 6.3 l h^{-1}. The CR equation developed for bottom-cultivated oysters ($CR = 6.35 \pm 0.59 \times DTW^{0.78\,\pm\,0.08}$) was very similar to the one reported by Riisgard (1988): $CR = 6.79 \pm 1.41 \times DTW^{0.73\,\pm\,0.22}$.

For suspended oysters, a low exponent in the equation $CR = 4.34 \pm 0.32 \times DTW^{0.41\,\pm\,0.08}$ indicated that CR increases relative to body size were lower than in bottom cultured oysters. Filgueira et al. (2008) reported that low exponents in CR power functions are expected when the condition index increases with body size. In keeping with this information, a significant and linear correlation between CI and DTW was found for the suspended category only ($r = 0.81$, $P < 0.001$, not shown in results). The low exponent is also consistent with the observation that gill size per unit DTW was relatively low in large suspended oysters. Gills in oysters not only serve in respiration; they contain cilia that create complex water flow patterns to capture food particles and transport them to the mouth (Newell and Langdon, 1996). It seems that suspended oysters direct most of their dietary supplement, derived from the high food flux environment in which they reside (Comeau et al., 2010), towards the buildup of energy stores and somatic growth, without proportional investments in gill development. My results predict that a 1 g DTW suspended oyster has a CR of 4.3 l h^{-1}. This rate is very similar to the 4.0 l h^{-1} g^{-1} reported by Pernet et al. (2007). Oysters in their laboratory investigation also originated from suspension culture.

4.3. Clearance rate (per unit area)

CR_{Area} in the most active floating bag leases was 66.5 ± 8.5 l h^{-1} m^{-2}. Depending on oyster size class distribution, this estimate could range from 41.3 ± 5.2 l h^{-1} m^{-2} (all bags contain oysters < 31 mm) to 83.5 ± 10.6 l h^{-1} m^{-2} (all bags contain oysters >65 mm). The elevated exponent value in the CR–DTW equation for bottom oysters would have amplified, incorrectly, this upper range estimate by 73% (i.e., from 83.5 ± 10.6 to 144.4 ± 18.3 l h^{-1}). Regardless of the calculation protocol, it is concluded that suspended oysters process small volumes of water compared to healthy natural reefs. Ermgassen et al. (2013) estimated the historical (ca. 1880–1910) CR_{Area} for natural reefs in 13 US estuaries. They reported rates up to 574 l h^{-1} m^{-2}. The estimate for the most highly-populated natural reefs in eastern NB was 901.3 ± 435.5 l h^{-1} m^{-2} (this study).

4.4. Bay-scale impact

The dataset was sufficiently detailed to conduct a bay-scale impact assessment in Foxley River, an intensive culture site where leases cover 22% of the bay area (1354 ha). An index of seston depletion ($I_D = CT/RT$) was calculated following Dame (1996). I_D provides an indication of how important seston uptake may be in relation to estuarine volume and tidal flushing. Clearance time (CT), the number of days required for the combined bottom and suspended stocks to filter the total estuarine volume (22.24×10^6 m^3), was estimated at 9.8 days. This estimate takes into account a total standing stock of 1095 t distributed among 32 suspended (100 ha) and 98 bottom (196 ha) leases. The estimate assumes that oysters were feeding 68.6% of the time (Comeau et al., 2012). Residence time (RT) is the number of days required for tidal action to replace the total estuarine volume. The deployment of a tidal gauge in 2012 and calculation of the tidal prism indicated an RT of 2.1 and 4.6 days during spring and neap tides, respectively (Thomas Guyondet, DFO, pers. comm., 2013). In keeping with these values, the I_D (CT/RT) estimate for Foxley River ranges between 2.13 (neap tide) and 4.65 (spring tide), meaning that the bay-scale food renewal rate by tidal action is on average 3.39 times faster than the filtration rate by cultivated oysters. Converting all bottom leases into suspended leases would increase the I_D to 4.94, consequently reducing the grazing pressure in the system. All of these I_D estimates fall in the upper range of I_D values reported for 11 other aquaculture bays (Dame and Prins, 1997). In intensive culture areas, it was found that grazing pressure has exceeded water renewal rates ($I_D < 1$). Such is the case for oyster table culture (I_D ~ 0.38) in Marennes-Oléron France (Dame and Prins, 1997), mussel raft culture (I_D ~ 0.54) in the Ria de Arosa Spain (Dame and Prins, 1997), and longline mussel culture (I_D ~ 0.34) in Tracadie Bay, PEI (Comeau et al., 2008). There is also evidence that bivalves naturally exerted a dominant effect in some coastal systems prior to the development of aquaculture. Historical (c. 1880–1910) baselines for North American oyster reefs suggest I_D values ≤ 1 for six of eight estuaries in the Gulf of Mexico (Ermgassen et al., 2013). In Foxley River, raising the suspended lease coverage from 22 to 100% of the bay area would reduce the I_D from its present 3.39 value to 1.17. Although these calculations do not take into account natural reefs, which are poorly documented, they suggest that cultivated oysters do not exert a dominant top-down control on phytoplankton abundance in the Foxley River system.

Finally, as the industry embraces suspended culture, coastal residents and recreational boaters tend to oppose the technique on the basis of visual or leisure amenity values. Others oppose suspended culture on the basis of perceived negative environmental impacts. At first glance, multiple floating structures distributed over large estuarine areas seem disruptive to ecological health. Yet, often overlooked are the positive ecological effects of suspended oyster culture. By making available a 3-dimensional substrate, suspended structures provide habitat for native fish and invertebrate species (DeAlteris et al., 2004; Marenghi and Ozbay, 2010; O'Beirn et al., 2004; Tallman and Forrester, 2007). Moreover, floating bags or cages in Atlantic Canada contain native oysters that were historically thriving in pre-colonial times (Kirby, 2004; Kirby and Miller, 2005), but have since been decimated by disease, overfishing, and deteriorating bottom habitats. There is compelling evidence that oysters improve estuarine water quality by filtering suspended particulate matter from the water column (Forrest et al., 2009); they may also serve as a top-down control of phytoplankton blooms in eutrophic systems. Such positive services are vital to the ecological integrity of estuaries and provide the impetus for oyster restoration projects worldwide.

5. Conclusion

This investigation provides one of the first numerical assessments of suspended oyster culture in eastern Canada. It was found that the present transition from bottom to suspended culture results in an actual reduction in oyster stocking density. Moreover, it was reported that suspended oysters have a weak grazing potential per unit body weight when compared to bottom oysters. A bay-scale assessment of an intensive culture site led to the conclusion that cultivated oysters do not exert a dominant top-down control on phytoplankton abundance.

Acknowledgments

The author gratefully acknowledges Rémi Sonier, Claire Carver, Thomas Guyondet and Lori Cuddy for input and advice at various stages of the project. Thanks are also due to oyster growers in Foxley River for their cooperation and assistance in the field, and to Tina Sonier for laboratory analyses. Suspended lease surveys were led by the New Brunswick Department of Agriculture, Aquaculture and Fisheries (NBDAAF) and the Atlantic Veterinary College (AVC), and more specifically by Sylvio Doiron, Jeffrey Davidson, Marie-Josée Maillet, Garth Arsenault and John Davidson. This project was funded by DFO's Program for Aquaculture Regulatory Research (PARR project 2012-G-05).

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Aquacultural Engineering 40 (2009) 87–94

Contents lists available at ScienceDirect

Aquacultural Engineering

journal homepage: www.elsevier.com/locate/aqua-online

Deterring coastal birds from roosting on oyster culture gear in eastern New Brunswick, Canada

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ARTICLE INFO

Article history:
Received 20 August 2008
Accepted 17 November 2008

Keywords:
Crassostrea virginica
Oyster
Aquaculture
Phalacrocorax auritus
Cormorant
Roosting
Birds

ABSTRACT

An ornithological survey was conducted along the eastern coastline of New Brunswick, Canada, where oysters are cultivated in suspension using PVC bags and wire-mesh cages. Thirteen bird species and a variety of unidentified shorebirds were observed roosting on the floating oyster gear. The double-crested cormorant (*Phalacrocorax auritus*) was the most common species observed (47.6% of all counts), closely followed by herring gulls (*Larus argentatus*) and common terns (*Sterna hirundo*) at 18.7% and 13.0%, respectively. Birds were densely aggregated where few cages or bags had been deployed. A gear-type effect was also detected: birds were more abundant on floating cages (mean = 47.9/100 m^2 of exposed area, S.E. = 5.8) than on floating bags (mean = 32.8/100 m^2, S.E. = 1.9). The survey was followed by two experiments designed to test the effects of gear modifications on bird abundance and diversity. For bags, results indicated that shallow immersion (~6 cm below surface) and floater instability were effective deterrents to *P. auritus*, reducing its abundance by a 37-fold factor. For wire-mesh cages, a dented triangular structure mounted on top of floaters was a harassing physical barrier to roosting behaviour, consequently reducing bird abundances to null (or near null) values.

1. Introduction

Communal roosting is a common behaviour in several species of social animals, including coastal birds (McGowan et al., 2006). This behaviour has evolved independently numerous times (Beauchamp, 1999; McGowan et al., 2006) and is thought to positively impact several species of seabirds (Roycroft et al., 2007) by enhancing the sharing of information (Ward and Zahavi, 1973; Ydenberg and Prins, 1984; Dall, 2002) and by promoting recruitment (Richner and Heeb, 1996; Dall, 2002). Roosting behaviour can also increase foraging efficiency, reduce predation risk and minimize thermoregulatory costs (Ydenberg and Prins, 1984; Beauchamp, 1999; McGowan et al., 2006). The behaviour has been studied extensively from an ecological perspective, providing a better understanding of roost choices (Luìs et al., 2001; Rogers, 2003; Rogers et al., 2006), species distribution (King, 1996; Bugoni and Vooren, 2005; Dittman et al., 2005) and hierarchical dominance in roosting populations.

From an aquaculture perspective, however, communal roosting is considered a nuisance. Birds predate on cultured fish stocks (Jenkins and Smith, 1998; Dorr et al., 2004; King, 2005), and their presence also raises other concerns, such as water contamination by faecal coliforms (Kirschner et al., 2004; Kuntz et al., 2004; Bucio et al., 2006), propagation of pathogenic agents (Flowers et al., 2004; Overstreet and Curran, 2004; Mitchell et al., 2005), and organic enrichment of sediments (Powell et al., 1991). Several bird-deterring techniques have been suggested in the literature (see review by Mott and Boyd, 1995). These methods include scaring effigies (Stickley et al., 1995; Seamans, 2004), repelling chemicals (Cotterill et al., 2004; McWilliam and Cheke, 2004; Harpaz and Clark, 2006), fencing and netting (Mott and Flynt, 1995; Nemtzov and Olsvig-Whittaker, 2003), harassment devices (Mott et al., 1998; Tobin et al., 2002), and the more-drastic solution of hunting (Bechard and Marquez-Reyes, 2003; McWilliam and Cheke, 2004).

In New Brunswick, Canada, oyster (*Crassostrea virginica*) farming is carried out in approximately 15 embayments (Fig. 1). Suspended culture, in which oysters are held inside floating PVC bags or floating cages (Fig. 2), is the predominant farming technique. In winter floaters are removed to lower stocks onto the bottom where they are protected from the thick ice. At other times, however, stocks are suspended at the surface in a relatively

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doi:10.1016/j.aquaeng.2008.11.003

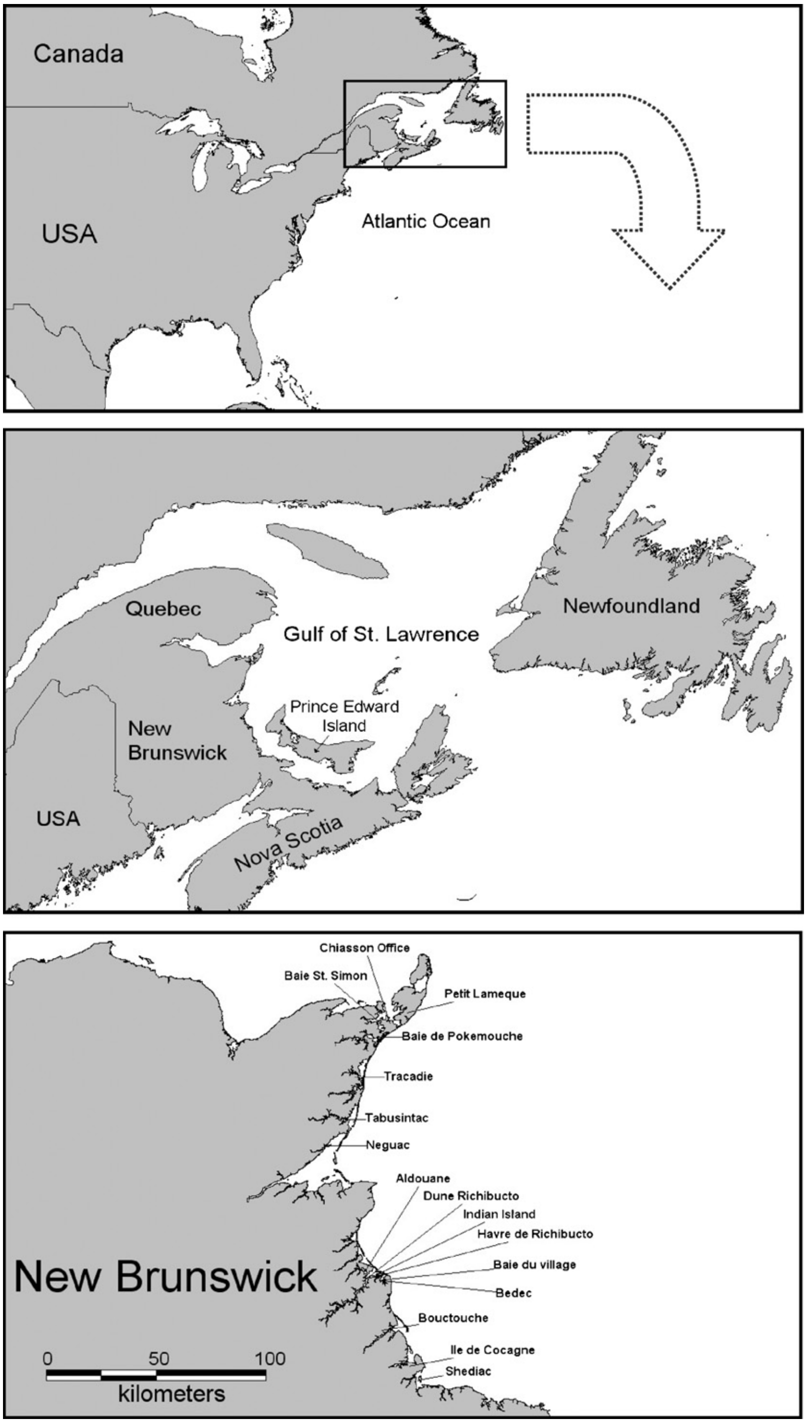

Fig. 1. Map of study area showing oyster farming sites in New Brunswick.

warm and phytoplankton-abundant environment, thereby enhancing shell growth and shortening the production cycle. When at the surface, stocks are easily accessible to growers for harvesting and grading procedures; moreover, the suspended bags or cages can be flipped and temporarily exposed to air, thereby desiccating biofoulers. The entire technique for suspending and flipping bags and cages has been developed in New Brunswick in the late 1990s.

Floating gear, on the other hand, provides substantial roosting areas for coastal birds. In 2004, Canadian food safety and fisheries agencies have requested that all oysters contained in floating bags or cages be depurated prior to harvest. The precautionary depuration procedure requires the transfer of suspended stocks onto the bottom 30 days prior to harvest (14 days if stocks are subsequently tested for coliforms as required by the Canadian Shellfish Sanitation Program, 2005). The new regulation increases both labour and time needed to complete the production cycle. Consequently, there is a growing interest in developing new floating gear designs that could prevent birds from roosting in oyster farms. The underlying rational is that effective bird-deterring designs would ultimately be exempted from the regulation pertaining to depuration.

In this paper, we begin by reporting results from an ornithological survey conducted in NB oyster farms. We identify bird species and report on their abundance in relation to current

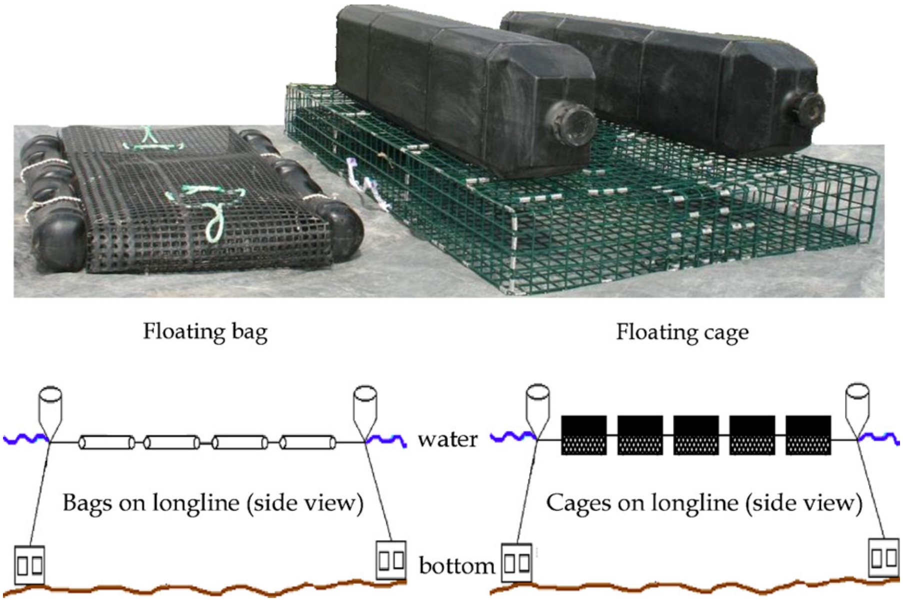

Fig. 2. Floating gear types currently present in New Brunswick oyster farms. The trademark for the floating cage is OysterGro®, manufactured by Bouctouche Bay Industries Ltd. in New Brunswick.

floating gear designs. The survey was followed by two field experiments examining the effects of gear modifications on bird abundance and diversity.

2. Methods

2.1. Survey

In September and October 2005, 15 embayments along the eastern coastline of New Brunswick were visited. The location of each embayment is identified in Fig. 1. A total of 22,600 floating bags and 4609 floating cages were examined for the presence of birds. Bird observations were carried out either from land or from a kayak using binoculars and a spotting scope. Only birds perching on oyster gear and any associated buoys were identified and counted. Bird counts reflect the maximum number of individuals seen at any one time during a disturbance-free period of 15 min. All counts were carried out between 10 a.m. and 4 p.m. A subsequent analysis indicated that the time of day had no significant effect on bird counts.

It is also important to note that the surface areas available for roosting differ according to gear type. For a floating bag, which has two small floaters and one PVC bag that are exposed and available for roosting, this area is approximately 0.35 m^2. For a floating cage, the available roosting area provided by the two large floaters is 0.45 m^2; the wire-mesh cage itself is too deep—12 cm below surface—to provide a roosting surface. In keeping with this information, counts were standardized as the number of birds per 100 m^2 of available (exposed) roosting area.

2.2. Floating bag experiment

In 2006, three different types of floating bags were deployed within an experimental setting. The first type consisted of standard bags (S) with lateral floaters typical of those in current use by most growers (Fig. 3a). For the second type, the S configuration was modified by positioning the two side floaters onto the top of the bag, thereby allowing the bag to sink approximately 3 cm below the water surface (Fig. 3b). While the two floaters remain a potential perching platform for birds, the area they offer to birds is less than 20% that of the unmodified bag. This modification was termed M1. In a second modification (M2), the S configuration was modified by positioning the two side floaters on top of the bag, but the bag itself was lowered approximately 6 cm below the surface using loose rope (Fig. 3c). The loose rope between the floaters and the bag rendered the floaters unstable.

The experimental bags were deployed in three embayments: Chiasson Office, Néguac, and Richibucto (see Fig. 1). At each of these sites, three longlines were deployed equidistantly (6.1 m) as illustrated in Fig. 4. Each longline held 11 floating bags per type (S, M1 and M2), which were dispersed in groups of three bags (except at the end of the longline where space was lacking and where grouping was limited to two bags). Details regarding bag layout are of no consequence since the entire longline itself was considered the statistical unit. For that reason, a single bird count (per bag type) was performed for each experimental longline. Bird counts represent the maximum number of individuals seen at any one time during a 1-h period. Counts were standardized to the number of birds per 100 bags. Species richness represents the number of different species observed during the count period. All observations were carried out at bi-monthly intervals between August 28 and November 7, 2006.

Data were partitioned into five 2-week intervals. Factors for each variable were analysed using a complete randomized block design with repeated measures according to gear type (fixed between-subjects factor with three levels [M1, M2 and S]), sites (fixed between-subjects factor with three levels [Chiasson Office, Néguac, and Richibucto]), sampling time (random factor with five levels of repeated measurements) and all their mutual interactions. Mauchly's test ($\alpha = 0.05$) was used to assess whether datasets conformed to the sphericity assumption required for a repeated measure analysis. When the sphericity assumption was not met, the degrees of freedom were adjusted accordingly using the Huynh–Feldt correction. Significant differences between all possible combinations of sample means for gear type were also assessed using Tukey's HSD test ($\alpha = 0.05$). All analyses were performed with SPSS 10.0 for Windows© (SPSS Inc., Chicago, IL, USA).

2.3. Floating cage experiment

Bouctouche Bay Industries Ltd. has developed the AntiCormo (AC), a bird-deterrent structure that can be fitted easily onto

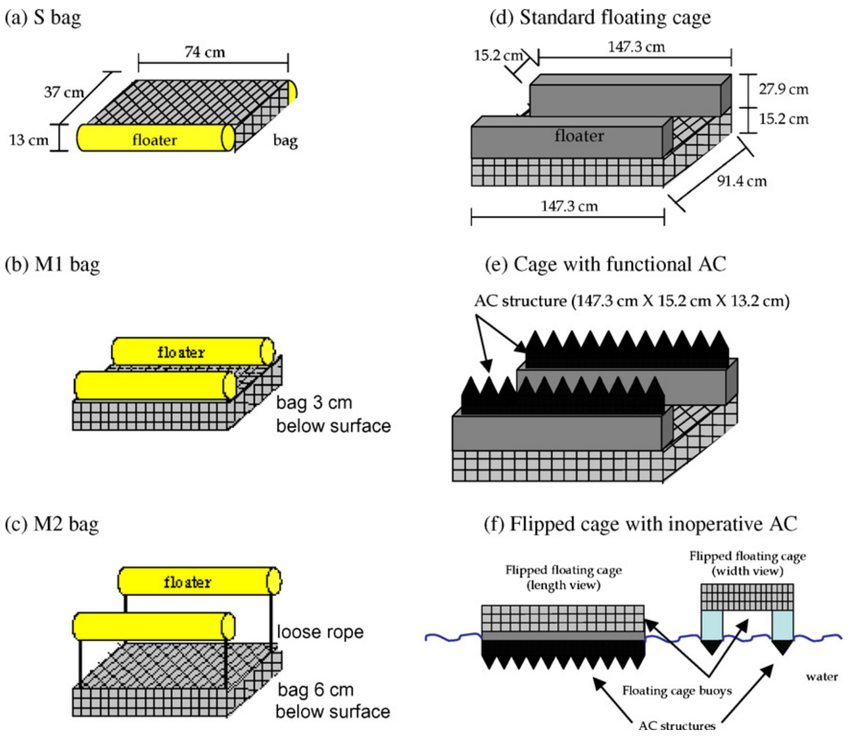

Fig. 3. Floating gear prototypes tested in the present study.

floating cages as illustrated in Fig. 3e. Taking into account the ability of birds to adapt over time, our goal in this experiment was to assess the effectiveness of the AC in deterring birds over an extended period. Our experimental approach was based on the full conversion of two farms and monitoring of bird abundance over an 18-week period (July 1 to November 3, 2007). One farm was located in Shediac Bay and contained a total of 140 floating cages (14 longlines of 10 cages); the second farm was located in Bouctouche Bay and held 100 cages (10 longlines of 10 cages). The two farms were selected because they were isolated, with the closest commercial aquaculture activities located at a distance of 5–10 km. The presence of bird colonies near the experimental farms was verified using two approaches: (1) by removing the AC devices at the start and completion of the experiment, and (2) by occasionally flipping cages and rendering the AC non-functional (i.e., underwater, as illustrated in Fig. 3f).

Once a week, bird counts were performed from land using a spotting scope. Counts were limited to four randomly selected longlines (the statistical unit). The count period consisted of four consecutive 15-min intervals. Data corresponding to the interval with the maximum number of individuals of the same species were kept for analysis. Descriptive statistics, including the standardized bird abundance per 100 floating cages, were computed for each experimental site.

3. Results

3.1. Survey

Thirteen bird species and a variety of unidentified shorebirds were observed roosting on floating oyster gear (Table 1). The most common species was the double-crested cormorant (*Phalacrocorax auritus*), representing almost half (47.6%) of all counts. Behaviourally, *P. auritus* was observed perching and preening, as well as drying its wings. Herring gulls (*Larus argentatus*) and common terns (*Sterna hirundo*) were also often spotted (18.7% and 13.0% of all counts).

Bird abundance was inversely correlated with the total roosting area made available by the floating gear (Fig. 5). The highest abundances, indicating a high degree of aggregation, were recorded at sites containing relatively few bags or cages. In keeping with these results, bird abundances were normalized to remove the effect of available roosting area. Following this correction, we found that birds were more abundant on floating

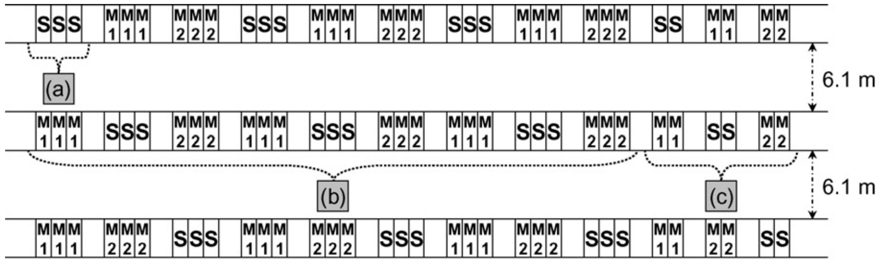

Fig. 4. Example layout of three experimental longlines holding three types of floating oyster bags: standard (S), first modification (M1) and second modification (M2). (a) Group of three bags of the same gear type. (b) Nine groups of three bags laid out in a random order. (c) End of experimental culture line with three groups of two bags laid out in a random order.

Table 1
Bird species surveyed at commercial oyster farming sites in New Brunswick.

Common name	Latin name	Total counts	% Total count
Double-crested cormorant	*Phalacrocorax auritus*	1588	47.6
Herring gull	*Larus argentatus*	624	18.7
Common tern	*Sterna hirundo*	435	13.0
Black-bellied plover	*Pluvialis squatarola*	160	4.8
Dunlin	*Caldris alpina*	146	4.4
Greater black-backed gull	*Larus marinus*	117	3.5
Immature gull	*Larus* spp.	70	2.1
Bonaparte's gull	*Larus philadelphia*	52	1.6
Ringed-billed gull	*Larus delawarensis*	51	1.5
Shorebirds spp.	*Caldris* spp.	28	0.8
Red-breasted merganser	*Mergus serrator*	23	0.7
Black duck	*Anas rubripes*	9	0.3
Greater yellowlegs	*Tringa melanoleuca*	8	0.2
Great blue heron	*Ardea herodias*	6	0.2
Lesser yellowlegs	*Tringa flavipes*	1	0.0
All species		3318	100.0

cages (mean = 47.9/100 m^2, S.E. = 5.8) than on floating bags (mean = 32.8/100 m^2, S.E. = 1.9) ($P < 0.001$, Mann–Whitney).

3.2. Floating bag experiment

Table 2 summarizes the outcome of the floating bag experiment. Gear type, which is the key factor of interest in the present investigation, was the only factor that yielded a significant effect on species richness in the main effects category. Moreover, gear type showed no interactions with sampling time, indicating that the effect on species richness was consistent through time. Similar effects were detected on bird abundance. Gear type exerted a significant influence on abundance and there was no interaction between gear type and sampling time. The same outcome was obtained whether all species were grouped or *P. auritus* was analysed separately. By contrast, no gear effect was found when analysing other species (e.g., *Larus* spp.) separately.

Fig. 6 shows mean species richness (panel A) and abundance (panel B) in relation to gear type. Post hoc analyses (Tukey's HSD) indicated that gear-type effects were attributable to differences between S and M2 bags. On average, species richness for S bags was approximately seven times higher than for M2 bags. A total of nine species were seen roosting on S bags throughout the duration of the experiment compared to only two species for M2 bags (*P. auritus* and *Larus marinus*). S bags also attracted a greater number of *P. auritus* compared to M2 bags. Average standardized abundance of *P. auritus* was approximately 37 times greater on S bags than on M2 bags.

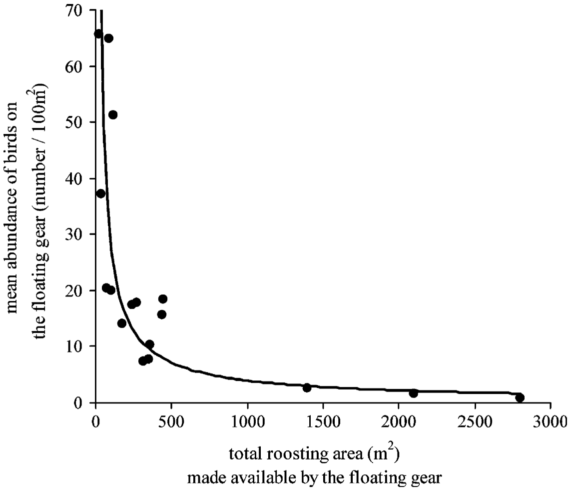

Fig. 5. Relationship between bird abundance and the roosting area made available by floating oyster gear. The solid line is a power-fit to the following function: $y = 1388.5x^{-0.8487}$ ($r^2 = 0.82$, $P < 0.001$). Data points represent the mean values of several sampling dates for individual sites.

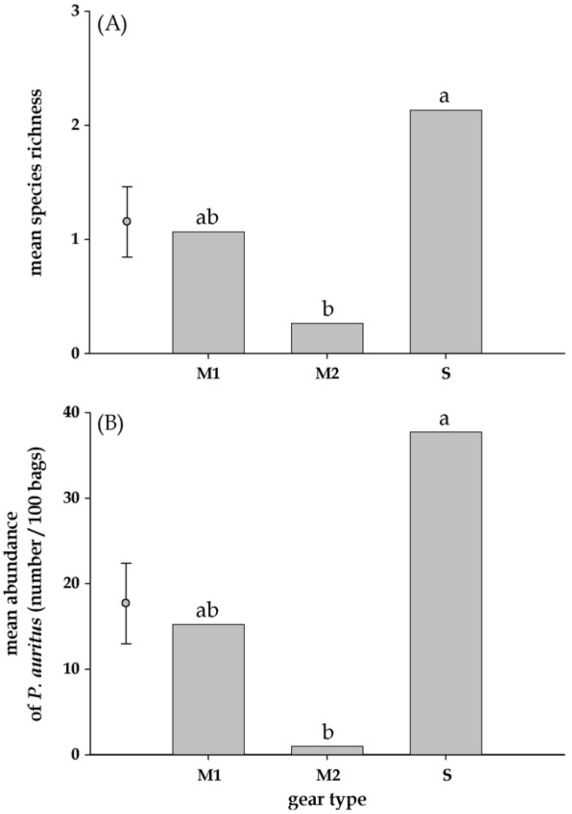

Fig. 6. Mean species richness (a) and abundance (b) of birds roosting on floating bags of type S (standard), M1 (first modification) and M2 (second modification). Means are presented with a single error bar provided by the standard model error (SME). Bars with different letters differ significantly from each other (Tukey's HSD).

3.3. Floating cage experiment

Fig. 7 shows the abundance time-series for the two experimental sites. At the Shediac site, AC structures were absent on several occasions (weeks 1–4, 15, 17 and 18); during these periods, abundance varied between 100 and nearly 500 birds per 100 floating cages. Abundance was also elevated at times when AC devices were inoperative due to the flipping of cages. Similar results were obtained at the Bouctouche site, with the exception that no birds were spotted in week 4 when the AC structures were absent. A total of 2195 individuals and 5 species (*P. auritus*, *L. argentatus*, *L. marinus*, *S. hirundo* and *Ardea herodias*) were identified at the two experimental sites; *P. auritus* was the dominant species, accounting for nearly 85% of all counts; *S. hirundo* and *L. argentatus* were also regularly spotted, with each species accounting for approximately 7% of all counts. Together these observations indicate that the two experimental sites were appropriate for testing the AC device.

Floating cages equipped with functioning AC devices attracted fewer birds. Weekly abundance estimates varied between 0 and 1.25 (S.E. = 0.72) birds per 100 floating cages at the Shediac site. During a 13-week period, only two individual birds, one *P. auritus* and one *L. argentatus*, were seen at this site. A total of 146 birds were spotted at the Bouctouche site, and weekly abundance

Table 2

Summary of a complete randomized block with repeated measures carried out on four variables: species richness, abundance of all species, abundance of *P. auritus*, and abundance of *Larus* spp.

Source of variation	d.f.	SS	Adjusted d.f.[a]	MS[b]	*F*	P[c]
(a) Species richness						
Site (S)	2	5.91	–	2.96	2.08	0.241
Gear type (G)	1	26.31	–	13.16	9.25	**0.032**
Between-subjects error	4	5.69	–	1.42	–	–
Time of sampling (T)	4	1.47	–	0.37	0.82	0.534
T × G	8	0.80	–	0.10	0.22	0.981
T × S	8	6.53	–	0.82	1.82	0.148
Within-subjects error	16	7.20	–	0.45	–	–
Total	44	53.91	–	–	–	–
(b) Abundance all species						
Site (S)	2	6019.38		3009.69	4.38	0.098
Gear type (G)	2	14181.00	–	7090.50	10.32	**0.026**
Between-subjects error	4	2745.84	–	686.46	–	–
Time of sampling (T)	4	4018.77	–	1004.69	5.33	**0.006**
T × G	8	2549.94	–	318.74	1.69	0.176
T × S	8	4730.53		591.31	3.13	**0.025**
Within-subjects error	16	3014.79	–	188.42	–	–
Total	44	37260.25	–	–	–	–
(c) Abundance of *P. auritus*						
Site (S)	2	4150.17	–	2075.08	4.00	0.111
Gear type (G)	2	10280.67	–	5140.33	9.91	**0.028**
Between-subjects error	4	2074.46	–	518.61	–	–
Time of sampling (T)	4	4878.04	3.24	1503.75	3.35	**0.050**
T × G	8	4317.36	6.49	665.46	1.48	0.257
T × S	8	8493.23	6.49	1309.10	2.92	**0.048**
Within-subjects error	16	5817.84	12.98	448.37	–	–
Total	44	40011.77	–	–	–	–
(d) Abundance of *Larus* spp.						
Site (S)	2	880.13	–	440.07	1.39	0.348
Gear type (G)	2	1846.76	–	923.38	2.92	0.165
Between-subjects error	4	1264.04	–	316.01	–	–
Time of sampling (T)	4	71.36	3.37	21.17	0.58	0.654
T × G	8	76.06	6.74	11.28	0.31	0.933
T × S	8	255.65	6.74	37.92	1.05	0.444
Within-subjects error	16	489.10	13.48	36.28	–	–
Total	44	4883.10	–	–	–	–

[a] Adjusted degrees of freedom (Huynh–Feldt correction) where the sphericity assumption is not met, $\alpha = 0.05$.

[b] Computed with adjusted degrees of freedom where available.

[c] Bold font indicates significance, $\alpha = 0.05$.

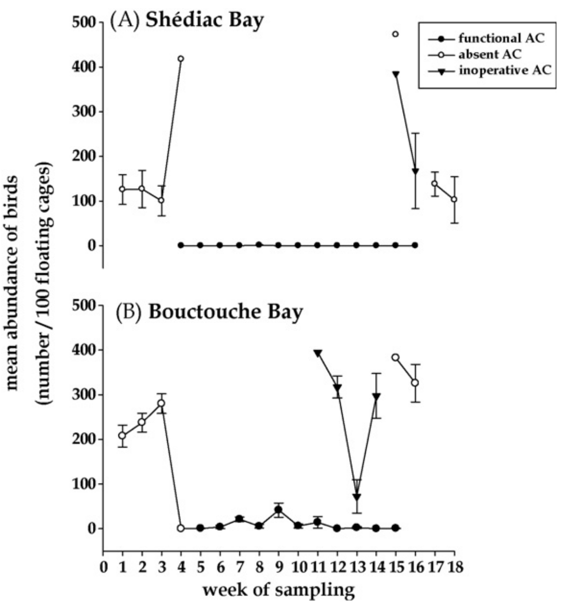

Fig. 7. Mean abundance (±S.E.) of birds roosting on experimental cages in Shediac (A) and Bouctouche (B). The time-series extends from July 1, 2007 (week 1) to November 3, 2007 (week 18).

estimates ranged from 0 to 41.9 (S.E. = 15.9) birds per 100 floating cages. There were no indications that birds progressively adapted (i.e., no increase in abundance over time) to AC devices at either site.

4. Discussion

4.1. Survey

An extensive ornithological survey indicated the presence of 13 bird species roosting on floating oyster gear along the eastern coastline of New Brunswick. Three species—*P. auritus*, *L. argentatus* and *S. hirundo*—which together were responsible for 79% of all counts, clearly dominated. These species have well-established breeding areas along the eastern coastline of New Brunswick (Erskine, 1992). They prey mainly on fish and small marine invertebrates such as zooplankton. There are previous reports on bird interference with aquaculture operations, although these studies have focused exclusively on predation of cultured stocks, such as *P. auritus* preying on farm-raised channel catfish (*Ictalurus punctatus*) in the southern United States (King, 1996). In our study, it was apparent that birds used floating oyster gear as roosting platforms.

Regarding abundances, the survey indicated that birds were densely aggregated where few culture units had been deployed (as per the relationship presented in Fig. 5). This result implies that the bird nuisance perception is function of the farming activity level. For example, in New Brunswick, the number of floating bags within individual leases varies from approximately 100 to 12,481 units (Comeau et al., 2006). We estimate, based upon the relationship shown in Fig. 5, that the lower end of activity (100 bags) could attract approximately 24 birds over a small body of water (~700 m^2), whereas the peak activity level (12,481 bags) may attract 49 birds dispersed over a much larger body of water (150,000 m^2). In keeping with this comparison, the amount of floating gear within a culture lease is perhaps a key parameter to consider when modelling the potential risks of water contamination by birds.

Another factor that seems relevant is gear type. Our survey suggests that birds have a preference for floating cages. This result may be attributable to cage design: compared to floating bags, floating cages are relatively stable and offer a large roosting area, attributes that are compatible with the large size and gregarious nature of *P. auritus* (Hatch and Weseloh, 1999). Also, floating cages provide an elevated platform (~17 cm above the waterline) compared to floating bags (~2 cm above the waterline). After diving, *P. auritus* usually looks for an elevated spot to perch, where it can spread its wings to dry its feathers (Hatch and Weseloh, 1999).

4.2. Floating bag experiment

The goal of this experiment was to compare bird diversity and abundance in relation to three bag deployment strategies: (1) standard (S) deployment, with the top portion of bags floating above surface; (2) modified (M1) deployment, with bags completely submerged ~3 cm under the surface; and (3) modified (M2) deployment, with bags submerged ~6 cm under the surface. Significant differences in bird diversity were found only between S and M2 bags; of the nine species observed on S bags during the experiment, only two (*P. auritus* and *L. marinus*) were seen on M2 bags. Three factors likely contributed to the decrease in diversity on M2 bags: depth (6 cm) at which the bag itself was maintained, floater instability, and interactions with floating organic debris. In terms of bag depth, it is noteworthy that both *P. auritus* and *L. marinus* have long tarsi, averaging approximately 8 and 9 cm, respectively (The New Brunswick Museum); species that avoided M2 have comparatively short tarsi. An influence of depth is consistent with reports of coastal birds changing their roost location with rising tides (e.g., Luìs et al., 2001; Rogers, 2003; Rosa et al., 2006). The M2 modification also increased floater instability. Our field notes indicate that the roosting time was very short (seconds) when *P. auritus* and *L. marinus* successfully landed on the M2 floaters; it was also noted that other species attempted to roost on M2 floaters but failed and immediately flew away. Lastly, S and M2 bags interacted differently with floating debris. S bags were often covered with common eelgrass (*Zostera marina*), which can be uprooted following storm events; M2 bags were generally free of this marine plant. This observation appears relevant because *S. hirundo* was occasionally seen feeding on small invertebrates entangled within *Z. marina*.

With respect to abundance, the total counts on S bags were dominated by *P. auritus* and *Larus* spp. The experiment showed that M2 bags attracted significantly fewer *P. auritus*. The reason(s) for M2 selecting against *P. auritus* cannot be determined with certainty. As indicated above, it is known that *P. auritus* has a marked preference for elevated perches where it can spread its wings to dry its feathers (Hatch and Weseloh, 1999). Floater instability and the depth of M2 bags probably prevented this behaviour. Gulls, on the other hand, do not exhibit this behaviour, which may explain why none of the experimental bag types significantly reduced the abundance of *Larus* spp.

4.2.1. Floating cages experiment

In this experiment, the effectiveness of a bird-deterrent device, the AntiCormo developed by Bouctouche Bay Industries Ltd., was evaluated at two sites over an 18-week period. The AC can be fitted onto existing floating cages as shown in Fig. 3e. In the absence of the AC device, floating cages generally attracted several birds as was expected from earlier survey results. This outcome indicates that local breeding populations, essential for the testing of the AC device, were present at the two experimental sites.

The AC device considerably reduced the number of birds roosting on floating cages at both experimental sites, with mean abundance falling from several hundred birds per 100 cages to null (or near null) values. Field notes indicate that the highest abundances at the Bouctouche site (e.g., mean of 41.9 birds/100 cages, week 9) were mainly associated with improperly installed AC devices. There were no indications that the birds adapted to properly installed AC devices. Therefore it appears that the AC was a harassing physical barrier, comparable to metal spikes or prongs commonly mounted, for example, on top of navigation buoys, park lights and gutters.

It is noteworthy that floating cages are occasionally flipped to control biofoulers as part of normal husbandry procedures. Once flipped, AC structures are submerged and the entire wire-mesh cage is exposed to air, thereby desiccating biofoulers. In our study, birds quickly resumed their roosting activities at times when cages were flipped. In New Brunswick, growers flip cages three to five times per year, and the desiccation of biofoulers normally occurs over 48 h, after which cages are returned to their normal position and the AC devices resume their full functionality. Evidently, cage flipping should be avoided some time prior to oyster harvesting. The “no-flip” period could be as short as 14 days in cases where there is follow-up testing for coliforms (Canadian Shellfish Sanitation Program, 2005).

5. Conclusion

This report presented possible mitigation measures to prevent the roosting of birds in oyster farms along the eastern coastline of New Brunswick. For floating bags, results suggested that floater instability coupled with an immersion depth of approximately 6 cm (for the bag itself) were effective deterrents to birds. Depth and floater instability were achieved simply by attaching loose ropes between floaters and bags. However, we recognize that this deployment scheme may not represent a practical option for the industry, given that bags must occasionally be flipped and exposed to air in order to control (desiccate) fouling organisms. Hence it is unlikely that the bag prototypes tested in the present investigation will be adopted by the industry. To date, no practical design has been found for floating bags, although the reported information on bird behaviour in the present report is useful for ongoing research.

For floating cages, a dented triangular structure (AC) mounted on top of each floater was an effective deterrent to birds. Moreover, from a practical perspective, the AC does not interfere with normal husbandry procedures. New floaters, commercially produced by Bouctouche Bay Industries Ltd. (New Brunswick, Canada), incorporate the AC (USA Patent No. D578,424 and Canadian Registration No. 125146).

Acknowledgements

Ève-Julie Arsenault, Tina Rousselle, and Roland Chiasson patiently identified and counted the birds; Alyre Chiasson, Rhéal Savoie, Marie-Josée Maillet, Marcel Léger, Christian Norris, Abel Noel, Sylvio Doiron, and Bettie Arsenault assisted in developing the project; Joe Caissie, Paul Savoie, Martin Mallet, and Serge Leblanc kindly provided access to the study sites. This study was funded by the Professional Shellfish Growers Association of New Brunswick in partnership with the Department of Fisheries and Oceans of Canada (Aquaculture Collaboration Research and Development Program, project MG-06-04-003) and the New Brunswick Department of Agriculture and Aquaculture.

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A comparative evaluation of the habitat value of shellfish aquaculture gear, submerged aquatic vegetation and a non-vegetated seabed

Article *in* Journal of Shellfish Research · December 2004

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Journal of Shellfish Research, Vol. 23, No. 3, 867–874, 2004.

A COMPARATIVE EVALUATION OF THE HABITAT VALUE OF SHELLFISH AQUACULTURE GEAR, SUBMERGED AQUATIC VEGETATION AND A NON-VEGETATED SEABED

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ABSTRACT The habitat value of modified rack and bag, shellfish aquaculture gear (SAG) used for the grow-out phase of the American oyster, *Crassostrea virginica*, submerged aquatic vegetation (SAV), *Zostera marina,* and a shallow nonvegetated seabed (NVSB) was comparatively evaluated over a 1-year period in Pt. Judith Pond, a tidal estuary in Southern Rhode Island. Enclosure gear was used to sample the three ecotypes, and organisms (>5 mm) were identified, enumerated, and measured to the nearest millimeter. Abundances of marine organisms and species diversity indices were used as measures of the habitat value of these ecotypes within each season. Environmental and geological parameters were not significantly different between the habitats. Emergent surface area (cm^2 m^{-2} of seabed) within each ecotype was estimated, and used to evaluate its role in providing habitat. The SAG habitat had a significantly greater surface area than either the SAV or NVSB habitats during all seasons. The physical structure of the SAG habitat protects juvenile fish from predators and provides substrate for sessile invertebrates that serve as forage for fish and invertebrates. The SAG habitat supported a significantly higher abundance of organisms per m^2 of seabed throughout the year. Species richness was also significantly greater in the SAG habitat compared with the SAV and NVSB habitats. A 2-way ANOVA indicated significant differences in species diversity (Shannon-Weiner index) between habitats. Tukey's HSD test indicated that the SAG habitat had significantly higher species diversity than the NVSB habitat, but no significant difference in species diversity was found between the SAG and SAV habitats. These findings indicate that shellfish aquaculture gear provides habitat for many organisms throughout the year, and is especially beneficial to ecosystems that support native species of recreationally and commercially important fish and invertebrates in their early life history stages. Therefore, we conclude that shellfish aquaculture gear has substantially greater habitat value than a shallow nonvegetated seabed, and has habitat value at least equal to and possibly superior to submerged aquatic vegetation.

KEY WORDS: shellfish aquaculture, habitat value, submerged aquatic vegetation

INTRODUCTION

Habitat is the place where an organism lives (Odum 1971). This simple definition is the basis for most ecologic studies involving habitat quality or value. Other considerations regarding the definition of habitat are that an organism at any particular life stage has only one habitat and that an organism's spatial distribution defines its habitat (Minello 1999). The characteristics of habitat that have been identified as being beneficial to organisms include physical structure, provision of food, substrate, hydrodynamics, and hydrology, and these must be specified to quantify habitat utilization by a particular species (Minello 1999). Physical structure is provided by submerged aquatic vegetation (SAV) or man-made structures like artificial reefs. The terms habitat "value" or "quality" when pertaining to fishery resources is defined as a habitat's ability to support a fishery resource (finfish, crustaceans, molluscs, and all other forms of marine animal and plant life). Studies that describe fishery resource habitat value primarily use species density or abundance data (Able, 1999). The purpose of this study is to comparatively evaluate the habitat value of modified rack and bag, shellfish aquaculture gear (SAG) used for the grow-out phase (Rheault & Rice 1995) of the American oyster, *Crassostrea virginica*, submerged aquatic vegetation (SAV), *Zostera marina,* and a shallow nonvegetated seabed (NVSB) over a 1-year period in Pt. Judith Pond, a tidal estuary in southern Rhode Island. The SAG habitat uniquely supplies an abundance of substrate due to the wire racks and rigid, plastic bags, in addition to the shell of the cultivated oyster.

In a study designed to estimate relative habitat value, Smith et al. (1989) used mark-recapture data and estimated densities of scallops (*Argopectin irradians*) to compare a recently transplanted eelgrass (*Zostera marina*) bed to a natural eelgrass bed. In a similar study, Fonseca et al. (1996) used abundances of shrimp, fish, and crab species to assess habitat value of the replanted eelgrass as compared with nonvegetated areas and naturally occurring eelgrass meadows. Recent studies involving oyster reefs have used similar criteria to determine relative habitat value by sampling nekton densities within the reefs. Coen et al. (1999a) conducted a long-term study comparing the habitat value of oyster reefs in the southeastern United States by measuring several parameters, including water quality and abundances, of resident and transient fauna. Faunal densities were used to compare species richness between natural and experimental reefs. Carr and Hixon (1997) compared fish assemblages and abundances to determine species richness on natural and artificial reefs. O'Beirn et al. (2001) investigated the organisms associated with oysters cultured in floating systems by measuring the number of macro-faunal species inhabiting these floating culture systems, so as to determine the species richness of this unique habitat.

Natural oyster reefs have been identified as essential fish habitat because not only do they support the oysters themselves but a myriad of other fishery resources. There is abundant evidence that indicates these reef communities are extremely diverse and show differences in species abundances as compared with adjacent nonvegetated, sand flat habitats. Oyster reef habitats are not only highly diverse but include species absent in adjacent soft-bottom environments (Coen et al. 1999b). In addition to obligate oyster reef residents, a variety of transient species occupy the reef in a facultative way (Posey et al. 1999). Grass shrimp, blue crabs, and other fish were observed utilizing the reefs possibly for foraging or refuge purposes. Breitburg and Miller (1998) reported that resident finfish populations are dependent on oyster reef habitats due to the physical extent of the reefs, their suitability as refuges from predators, and abundance of prey for consumption. These characteristics influence the abundance, growth, and reproduction of these resident finfish, thus demonstrating that oyster reefs enhance fish production. There is evidence that the 3-dimensional structure of oyster reefs affect the spatial distribution of various fish and perhaps the overall abundances. Striped bass and other predatory fish have been observed to hover near reefs utilizing them as foraging sites Breitburg 1999).

Habitats that exhibit structural complexity have been shown to support higher numbers of species as compared with barren non-vegetated bottom types (Orth & Heck 1980). Orth et al. (1984) concluded that an increase in habitat complexity due to eelgrass density should increase refuges for prey species. Man-made structures have also been shown to increase abundances of fishery resources (Carr & Hixon 1997). Man-made structures or "artificial reefs" may be specially constructed and consist of concrete rubble (Kelch et al. 1999) used for the purposes of creating habitat for fish. Grossman et al. (1997) hypothesized that if habitat is limiting, new artificial reefs can potentially increase fish production through 3 mechanisms: (1) an increase of foraging habitat for adult, juvenile, and/or newly recruited fishes; (2) an increase in breeding habitat; and (3) an increase in predator refuge or resting habitat. Therefore, shellfish aquaculture gear may serve as an artificial reef habitat by virtue of its inherent structural complexity and extensive time spent on the seafloor throughout the year, thereby increasing the fish production in the ecosystem.

MATERIALS AND METHODS

Study Area

Three habitats (SAG, SAV, and NVSB) were sampled in Pt. Judith Pond, Rhode Island, a shallow 6 km tidal estuary that discharges into Block Island Sound. The 1.0 h aquaculture lease site contained over 600 oyster cages, each consisting of a 1.8 m × 0.6 m × 0.6 m wire cage that held 12 mesh bags of shellfish on shelves. The oyster cages were placed 2.4–6.1 m apart on the seabed in 2.4–3.0 m of water. The SAV and NVSB habitats were located approximately 1.5 km south of the aquaculture lease in Pt. Judith Pond at similar depths of water.

Experimental Design

The research design was a four (season) by three (habitat type) factorial design with three replicates within each habitat. Three habitats (SAG, SAV, and NVSB) were seasonally sampled in replicate between December 2000 and October 2001 so as to evaluate the following habitat characteristics: macro-epibenthic fauna community structure, and the physical, chemical, and geological environmental conditions. All three habitats sampled using enclosure type gears to maximize the efficiency and consistency of sampling (Rozas & Minello 1997).

Field and Laboratory Methods

Moonstone Oyster Company cultivates the American Oyster (*Crassostrea virginica*) in cages that are cleaned every 4–6 months. We selected cages for sampling that had been cleaned 4–6 weeks prior to each seasonal sampling so that they would have a representative seasonal fouling population. Lift-nets (2.1 m × 0.9 m with a 2-mm mesh) were placed beneath three randomly selected SAG units 2 weeks before sampling to allow sufficient time for swimming organisms to return to the cages following the disturbance of lifting the cage to place the lift nets underneath.

A scuba diver deployed the lift-net so that it completely enclosed the oyster cage during recovery. All free swimming epifauna >5 mm were recovered from the lift net enclosure along with three randomly selected oyster bags, and were taken back to the laboratory for analysis. Each oyster cage was also randomly sampled in five locations with a 0.022 m^2 (15 cm × 15 cm) quadrat to assess sessile invertebrate growth. The oyster cages are constructed of 5.1 cm mesh, vinyl-coated, 2 mm diameter wire. Percent cover of each biofouling organism within each quadrat sample was assessed to the nearest class and/or phylum. The percent cover of sessile invertebrate growth on the oyster bags was determined in a similar fashion. Total biomass of sessile invertebrates on the cages and bags was estimated for the entire surface area of the cages and bags by extrapolating mean sample values to the total surface area. A random subsample of 10 oysters was taken from each of three bags taken from each cage. Oyster length and width was measured to the nearest millimeter using vernier calipers and the surface area of the oysters and sessile invertebrate growth on both sides was estimated to the nearest square centimeter. Results were averaged within seasons and extrapolated over an average of 200 oysters per bag or 2,400 oysters per cage. The total surface area and sessile invertebrate coverage (cm^2) for each oyster cage consisted of the sum of the surface area of the oyster cage, the 12 oyster bags, and the seasonal average surface area of the 2,400 oysters. These sums were divided by the area enclosed by the lift net used to sample the SAG habitat (1.95 m^2). Thus, surface area and invertebrate growth are referenced to area (m^2) of the seabed.

The SAV and NVSB habitats were sampled on the same day within a few hours of noontime during each of the seasons. These habitats were randomly sampled using a 2-mm mesh drop-net (2.13 m × 0.92 m) and a venturi-driven suction dredge deployed from a small skiff. The animals were collected in a 2 mm-mesh catch bag and returned to the laboratory for analysis. The emergent portion of the SAV habitat was randomly subsampled with a 0.25 m^2 quadrat (3 replicates) each season. The eelgrass blades within each quadrat were clipped at the base and measured to the nearest 100 cm using vernier calipers. Sessile invertebrate growth (cm^2) on the SAV was similarly estimated. The NVSB habitat was devoid of emergent substrate and attached sessile invertebrates.

All free swimming organisms >5 mm in length collected from each of the three habitats were identified to the genus and species, and measured to the nearest millimeter using vernier calipers. Temperature, salinity, and dissolved oxygen were seasonally measured during each sampling event in each habitat. Sediment from each habitat was collected seasonally using a 7.5-cm diameter ×15.2-cm deep corer. Mean sediment grain size was determined by dry sieve analysis (Folk 1968).

Data Analysis

Seasonal environmental parameters (temperature, salinity, and dissolved oxygen) were analyzed by 2-way analysis of variance (ANOVA) without replication (EXCEL 1997) between habitat and season. The environmental dependent variables for each season were also analyzed using 1-way analysis of variance (ANOVA). Tukey's honest significant difference (HSD) test was used to compare treatment means when an F-test indicated significant treatment effects (SPSS vs.10 1999). Sediment type data for each habitat was characterized according to percent gravel, sand, and silt-clay using a 2-way ANOVA without replication (EXCEL 1997) between habitat and season. This analysis was repeated after subtracting the gravel component from the oyster cage habitat to compensate for the presence of shell hash from the aquaculture operations. Physical habitat complexity was measured in terms of emergent surface area within each habitat. The average surface area within each of the replicates for each habitat was log transformed (ln(cm^2)) to satisfy the homogeneity of variance assumption for an analysis of variance (Zar 1984). The average surface area was compared with a 2-way ANOVA (SPSS vs.10 1999) between habitats and seasons, and Tukey's HSD test (SPSS vs.10 1999) was used to compare treatment means when an F-test indicated significant treatment effects.

The community structure was analyzed using Ecological Methodology (Krebs 1989) statistical software (Exeter Software 2000). The raw data used in the statistical software consisted of species abundances (3 replicates) within each habitat for each season. Species richness was determined by the Jackknife method for quadrat counts (Heltshe & Forrester 1983). Shannon-Weiner species diversity and Smith and Wilson species evenness indices were generated using Ecological Methodology statistical software (Exeter Software 2000). The indices of species richness, diversity, and evenness within each habitat were each analyzed using a 2-way ANOVA (SPSS vs.10 1999) between habitat and season. Tukey's HSD test (SPSS vs.10 1999) was used to compare treatment means when an F-test indicated significant treatment effects.

Species abundance data within each habitat were compiled into 5 categories for analysis; total abundances of all organisms sampled, fish, crustacean, mollusk abundances, and total surface covered by sessile invertebrates. The abundance data were log transformed (ln(X)) to satisfy the homogeneity of variances assumption (Zar 1984) and analyzed using a 2-way ANOVAs (SPSS vs.10 1999) between habitat and season for each abundance category. Tukey's HSD test (SPSS vs.10 1999) was used to compare treatment means when an F-test indicated significant treatment effects. Correlation analysis (EXCEL 1997) was used to investigate the relationship between the total abundance of animals observed in each habitat and season, and the emergent surface area found in each habitat and season.

RESULTS

Environmental Parameters and Sediment Characteristics

There were no significant differences in temperature, dissolved oxygen, or salinity between sites ($P < 0.05$) in any given season. Temperature varied seasonally from 3.0 to 23.7 °C; salinity was influenced by rainfall and ranged from 25.0 ppt to 34.6 ppt; and dissolved oxygen peaked in winter/spring at 11.9 mg/L and was lowest in spring/summer at 6.4 mg/L. The three sampling sites had a similar grain size composition, dominated by sand (mean 93.5%) and silt-clay (mean 6.5%), however there was a substantial gravel component (4.27%) in the SAG site that was comprised primarily of oyster shell fragments. After removing this fraction, the sediments from the three sites were not significantly different from each other ($P < 0.05$).

Habitat Structure

Habitat structure, described in terms of emergent surface area (cm^2) per m^2 of seabed, varied as a function of habitat type and season (Fig. 1). The log transformed average emergent surface area varied significantly both between sites and between seasons ($P < 0.001$). There were significant differences ($P < 0.05$) between each of the 3 habitats (SAG>SAV>NVSB), and significant differences ($P < 0.01$) between each of the seasons (except between spring/summer and winter/spring). The SAG habitat, due to the cages, bags, and oysters, provided an average of more than 60 times the emergent surface area per square meter over the course of the year than the SAV habitat. The SAV habitat had mean shoot densities of 554/m^2 in the spring/summer and summer/fall seasons and 224/m^2 in the fall/winter and winter/spring seasons. The NVSB habitat was devoid of emergent surface area during all seasons.

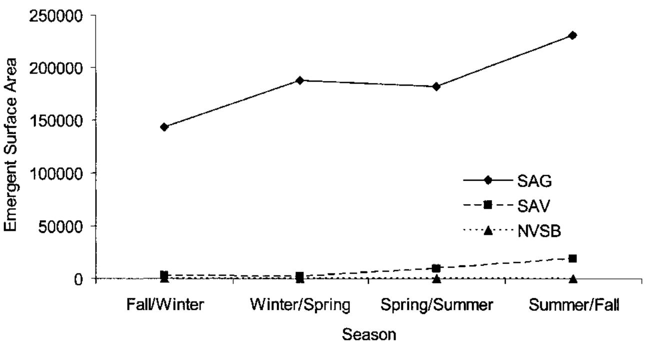

Figure 1. Emergent surface area (cm^2/m^2 of seabed) for each habitat and season.

Community Structure

Species richness was also consistently higher in the SAG habitat (Fig. 2a). There were significant differences ($P < 0.01$) between habitats, and between seasons ($P < 0.05$). Species richness was significantly different between each habitat (SAG>SAV>NVSB) and between fall/winter and summer/fall seasons. The mean Shannon-Weiner Index values of species diversity were highly significantly different between habitats ($P < 0.001$) and between

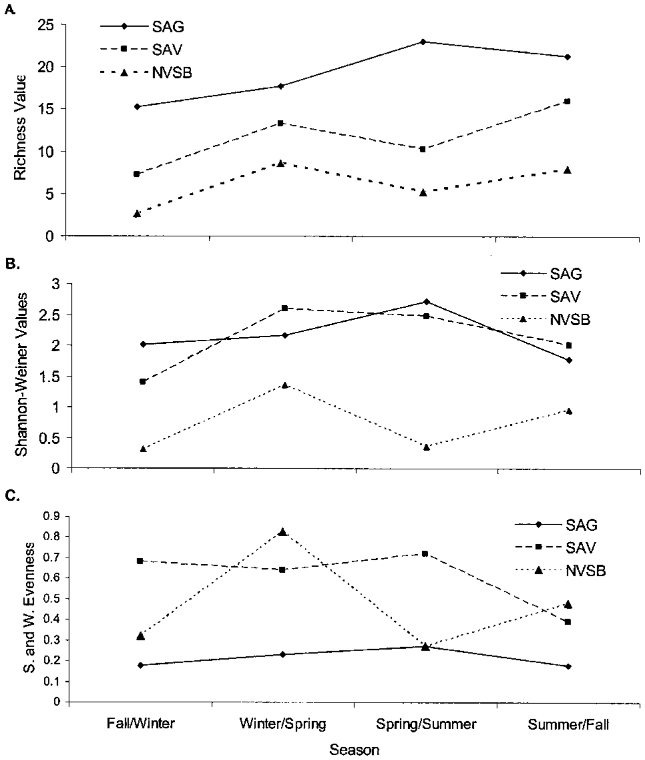

Figure 2. A. Species richness values for each habitat and season, B. Mean Shannon-Weiner values for each habitat and season, C. Mean Smith and Wilson measure of evenness values for each habitat and season.

seasons ($P < 0.01$). The SAG habitat was not significantly different from the SAV habitat ($P > 0.05$), however both of these habitats were highly significantly different ($P < 0.01$) from the NVSB (see Fig. 2b). Significant differences ($P < 0.05$) in species diversity were also found between the fall/winter and winter/spring sampling and between fall/winter and spring/summer. The SAG habitat showed consistently lower Smith and Wilson species evenness values than either the SAV or NVSB because a few species tended to dominate this habitat (see Fig. 2c). There were highly significant differences in species evenness between habitats ($P < 0.001$), but not between seasons ($P > 0.05$). The SAG habitat was significantly lower in species evenness than either the SAV or NVSB habitats ($P < 0.05$).

Species Abundances

The SAG habitat consistently supported far greater abundances of organisms than either the SAV or the NVSB habitats throughout the year (Fig. 3). There were highly significant differences ($P < 0.001$) between habitat and seasons for the species abundance data. There was a highly significant difference ($P < 0.001$) in species abundance between each habitat (SAG>SAV>NVSB). There was also a significant difference ($P < 0.05$) in species abundances between all seasons except winter/spring and spring/summer sampling periods showed no significant differences ($P > 0.05$). A strong correlation (r = 0.94) was found between the total abundance of organisms in each habitat and season and the emergent surface area available in corresponding habitat and season (Fig. 4).

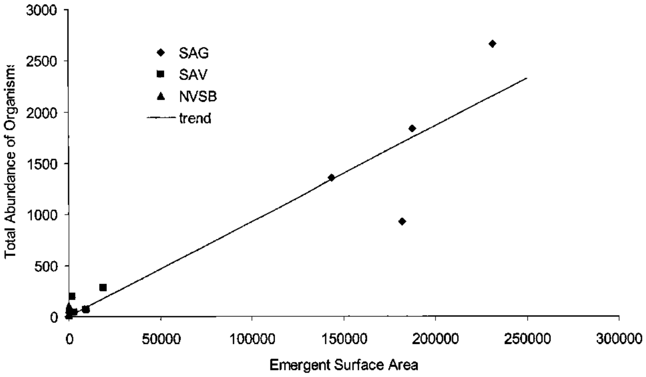

Figure 4. Correlation of total abundance of organisms (abundance) and emergent surface area (surface area cm^2/m^2 of seabed).

Ten fish species were identified inhabiting one or more of the three habitats sampled during the course of the study (Fig. 5), and individual fish species abundances are shown for each habitat and season in Figure 6. There were highly significant differences ($P < 0.001$) in fish abundances between habitats and seasons. The greatest fish abundances ($P < 0.01$) occurred in the SAG habitat followed by the SAV habitat and then the NVSB habitat. The summer/fall sampling period had significantly higher ($P < 0.01$) fish abundances compared with any other season. With two exceptions, the SAG habitat supported higher abundances of fish than either SAV or NVSB habitats. The Northern Pipefish (*Syngnathus fuscus*) in the spring/summer and summer/fall and the Winter Flounder (*Pleuronectes americanus*) in the summer/fall were unique to the SAV. There were many species of fish that were unique to SAG including the American eel (*Anguilla rostrata*), oyster toadfish (*Opsanus tau*), rock gunnel (*Pholis gunnellus*), and Atlantic tomcod (*Microgadus tomcod*). Several fish species were sampled throughout each season in the SAG, which included the seaboard goby (*Gobiosoma ginsburgi*), grubby (*Myoxocephalus aenaeus*), tautog (*Tautoga onitis*), and cunner (*Tautogalabrus adspersus*). The SAG habitat was the only habitat sampled that supported one or more fish species year-round.

Thirteen crustacean species were identified to inhabit one or more of the three habitats sampled during the course of the study (Fig. 7), and individual crustacean species abundances are shown for each habitat and season in Figure 8. There were highly significant differences ($P < 0.01$) in crustacean abundances between habitats and seasons. The greatest abundances occurred in the SAG habitat followed by the SAV habitat and then the NVSB habitat. The summer/fall sampling period had significantly higher ($P < 0.01$) crustacean abundances compared with any other season. The American Lobster, *Homarus americanus*, was the only crustacean unique to the SAG habitat (5 observed individuals). The average carapace length was 6.3 cm (S.E. ± 0.88), which places these lobsters in the juvenile phase of their lifecycle (Hudon 1987).

Seven mollusk species were identified to inhabit one or more of the three habitats sampled during the course of the study (Fig. 9), and individual mollusk species abundances are shown for each habitat and season in Figure 10. There were highly significant differences ($P < 0.01$) in mollusk abundances between habitats and seasons. The greatest abundances occurred in the SAG habitat followed by the SAV habitat and then the NVSB habitats. The winter/spring sampling period had significantly higher ($P < 0.01$) crustacean abundances compared with any other season.

Sessile invertebrate species were present in both SAG and SAV habitats (Fig. 11). The NVSB habitat was devoid of surface and hence the absence of sessile invertebrates. Statistics were not performed to detect differences between habitats due to the high variability of sessile invertebrate abundances.

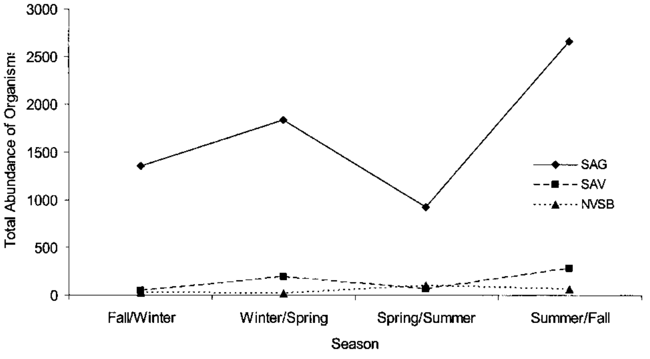

Figure 3. Total abundances of organisms collected within each habitat and season.

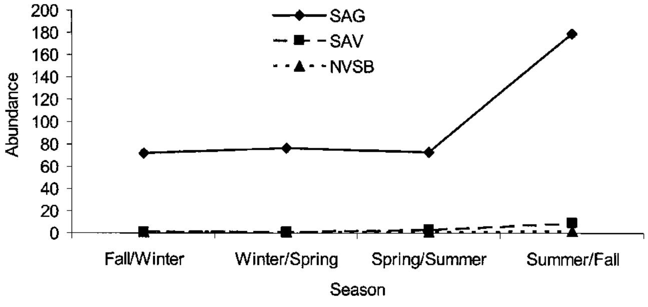

Figure 5. Total fish abundances found within each habitat and season.

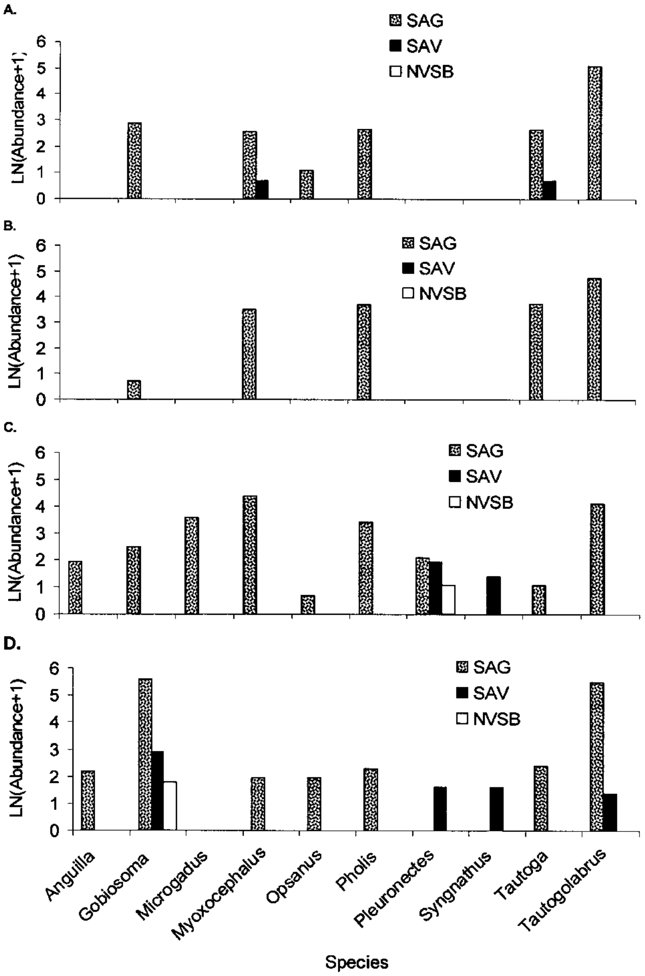

Figure 6. Total abundances (ln(abundance+1)) of individual fish species found within each habitat during: A. fall/winter, B. winter/spring, C. spring-summer, and D. summer-fall sampling periods for the following species: *Anguilla rostrata*, *Goliosoma spp.*, *Microgadus tomcod*, *Myoxocephalus aenaeus*, *Opsanus tau*, *Pholis gunnellus*, *Pleuronectes americanus*, *Syngnathus fuscus*, *Tautoga ontis*, *Tautogolabtrus adspersus*.

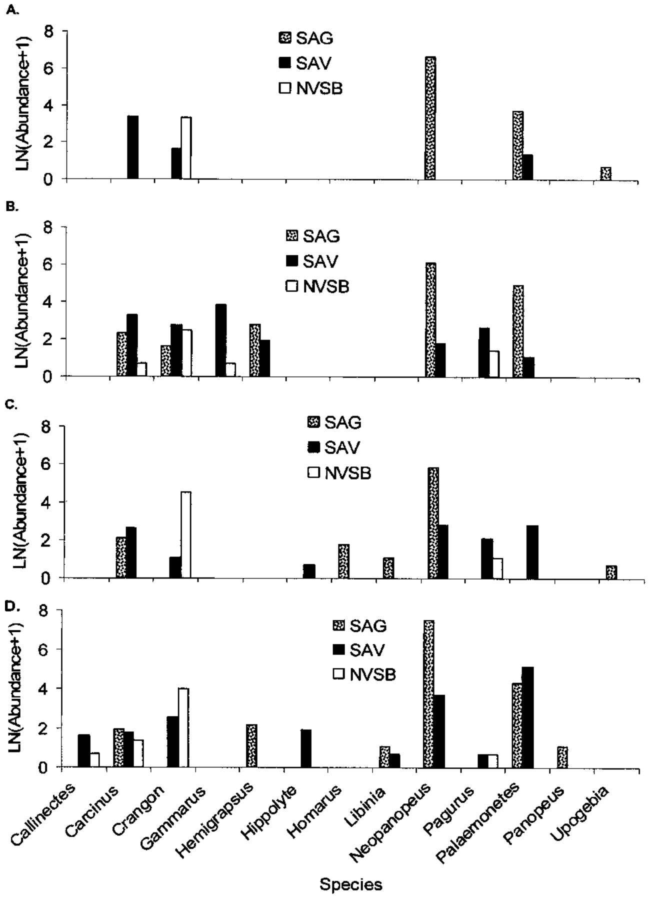

Figure 8. Total abundances (ln(abundance+1)) of individual crustacean species found within each habitat during: A. fall/winter, B. winter/spring, C. spring-summer, and D. summer-fall sampling periods for the following species: *Callinectes sapidus*, *Carinus maenus*, *Cragnon septemsoinosa*, *Gammarus spp.*, *Hemigrapsus sanguineus*, *Hippolyte spp.*, *Homarus americanus*, *Libinia emarginata*, *Dyspanopeus sayi*, *Pagurus longicarpus*, *Panopeus spp.*, *Upogebia affinis*.

DISCUSSION

Habitat is the place where an organism lives during any part of its lifecycle (Odum 1971). The ecologic value of habitat is inferred by quantifying the resident and transient marine organisms associated with a particular habitat. Consequently, the greater the abundance and diversity of fish in a particular habitat, the greater its habitat value (Able 1999). SAV and natural oyster reefs have been identified as important fish habitats not only because of shelter they provide to resident and transient marine organisms, but also because of the ecologic services they provide to the surrounding environment. The objective of our study is to comparatively evaluate the habitat value of SAG, SAV and NVSB in a small estuary. The SAV habitat sampled in this study is typical of other SAV

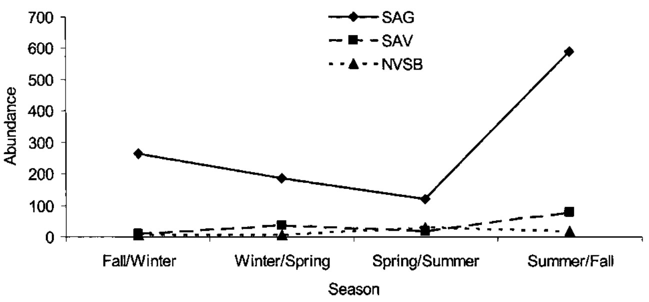

Figure 7. Total crustacean abundances found within each habitat and season.

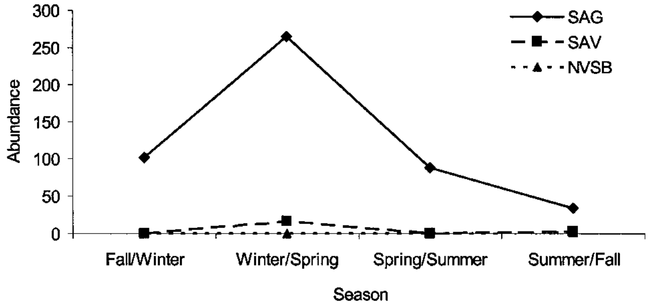

Figure 9. Total mollusk abundances found within each habitat and season.

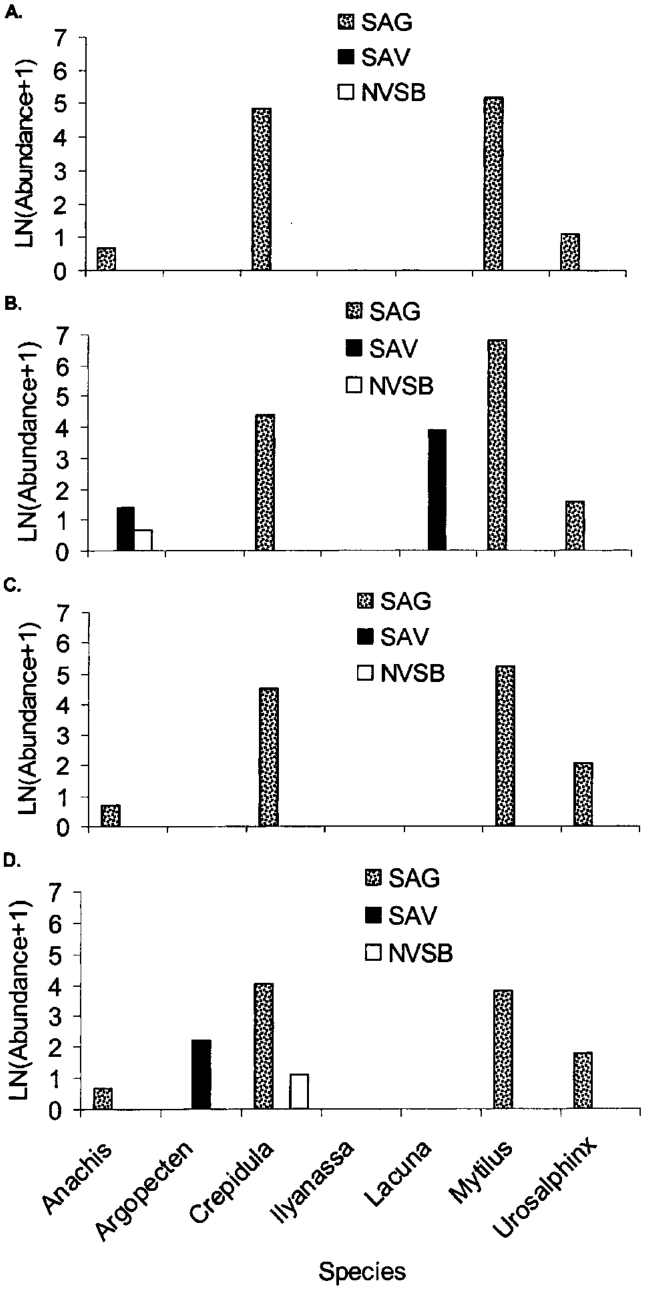

Figure 10. Total abundances (ln(abundance+1)) of individual mollusk species found within each habitat during: A. fall/winter, B. winter/spring, C. spring-summer, and D. summer-fall sampling periods for the following species: *Anachis spp.*, *Argopecten irradians*, *Crepidula fornicata*, *Ilyanassa trivattata*, *Lacuna vincta*, *Mytilus edulis*, *Ursalphinx cinerea.*

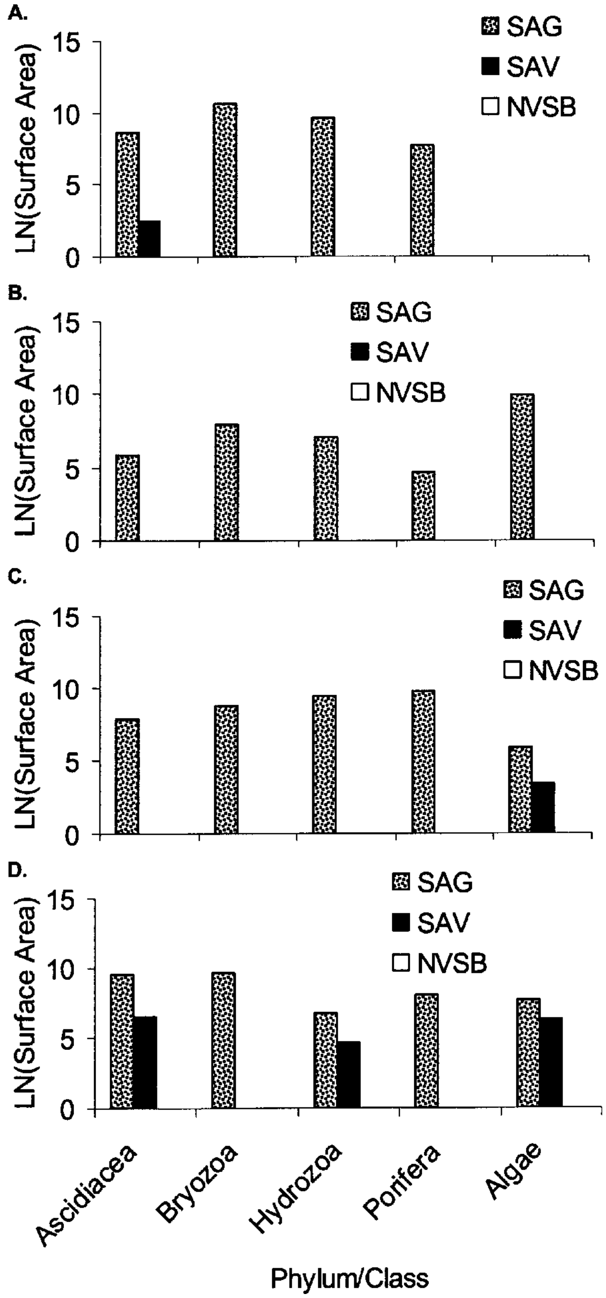

Figure 11. Total surface area (ln(surface area (cm^2)) of sessile species found within each habitat during: A. fall/winter, B. winter/spring, C. spring-summer, and D. summer-fall sampling periods for the following phylum/class groups: *Ascidiacea, Bryozoa, Hydrozoa, Porifera, Algae.*

habitats in New England and the mid-Atlantic regions based on eelgrass shoot density (Thayer et al. 1984),

The environmental parameters were relatively consistent among habitats within each season. No significant differences were observed between habitats for temperature, salinity, and dissolved oxygen, as was expected considering each habitat is contained within the same estuary. Also, as expected, the major differences among environmental parameters occurred between seasons. Sediment type between habitats was found to be similar after the gravel component was removed from the SAG site. The gravel component in the SAG site consisted of shell hash, which is a direct result of the aquaculture activities that take place over the seabed. The differences observed in species abundances and diversity between habitats are not likely to be related to environmental or geological parameters. Therefore, we believe that the observed differences in species composition and abundances are influenced by differences in habitat composition, structure, and complexity.

There was a highly significant difference in emergent surface area (cm^2) between each habitat that was strongly correlated with abundance of organisms observed. The NVSB habitat supported significantly fewer organisms than either SAV or the SAG habitats throughout the year. The SAV emergent surface varied throughout the year due to seasonal growth and mortality patterns. The SAG emergent surface area varied seasonally as a result of the measured changes in the surface area of the oysters, whereas the surface area of the cages and bags remained constant. We believe that the higher abundances of species found in the SAG habitat throughout the year are related to the high surface area, the large numbers of spaces inside the cages that serve as refuge, and the prevalence of fouling organisms and forage. Structural heterogeneity was not considered when quantifying each habitat. The SAG habitat is constructed of 2-inch (5.08 cm) plastic-coated wire mesh. It can be assumed that the size of the wire mesh restricted many of the predator species of certain sizes and hence the cages became a refuge for many of the juvenile species of fish. These results are consistent with many studies that have recognized increased habitat complexity supports higher abundances of organisms due to increased predator protection (Orth et al. 1984, Ryer 1988, Beck 2000).

The high surface area within the shellfish aquaculture gear provides habitat not only for mobile fauna but also sessile biofouling invertebrates. Sponges, hydroids, bryozoans, and ascidians were found in both the SAG and SAV but the SAG habitat clearly displayed larger abundances of sessile invertebrate species. The SAV does support epiphytic and sessile invertebrate growth but not to the extent of the SAG. Although not intensively studied in this research, sessile invertebrate communities form the base of the food web for many artificial reef communities (Blancher et al. 1994). The high prevalence of sessile invertebrate communities on the SAG not only increases habitat complexity, but also increases food resources for the marine organisms inhabiting the aquaculture gear.

The SAG habitat shares many attributes and similarities with natural oyster reefs and artificial reefs. The oysters within the aquaculture gear are providing many of the same ecologic services as those found within naturally occurring oyster reefs. These ecologic services include but are not limited to particle clearance, nutrient removal and remineralization, benthic-pelagic coupling, and the creation of refuge from predators (Coen et al. 1999a, Dame 1999). The SAG also provides 3-dimensional structural complexity and many of the same benefits that artificial reefs provide in areas where habitat is limiting. Studies have shown and suggested that biologic services of artificial reefs include foraging habitat and predator refugia to resident and transient marine organisms (Blancher et al. 1994, Bohnsack 1989).

The abundance (organisms >5mm) and species richness exhibited in the aquaculture gear was greater than the eelgrass habitat, which in turn was greater than the unvegetated site, consistent with previous studies (Orth & Heck 1980, Mattila et al. 1999, Heck et al. 1995). This research clearly indicates more organisms inhabit the SAG habitat either SAV or NVSB habitats per square meter of seabed throughout the year. Species diversity levels were similarly higher in the shellfish aquaculture gear and the eelgrass ecotypes than in the unvegetated bottom consistent with findings of Marshall-Adams (1976), Mattila et al. (1999), Heise & Bortone 1999). Average species diversity in the SAG habitat was higher, but not significantly, than in the SAV habitat. The evenness measures varied greatly for each habitat throughout the year, however the SAG habitat had consistently lower evenness than the other ecotypes because of the hyperdominance of several species within the aquaculture gear (*Dyspanopeus sayi, Tautogalabrus adspersus, and Mytilus edulis*). In contrast, the SAV habitat was rarely dominated by a few species, but rather supported a more equal distribution of organisms. The NVSB habitat showed a greater fluctuation of evenness values directly affected by the abundances of the sand shrimp (*Crangon septemspinosa*) sampled during each season. The sand shrimp was by far the most dominant species in the NVSB habitat and accounted for 87% of the NVSB organisms sampled throughout the year.

The abundance and species diversity data elucidate the similarities and differences between each of the three habitats. The oyster cages supported much greater species abundances than eelgrass, but displayed similar species diversity (as shown by the Shannon-Weiner index). Eelgrass is a habitat known to provide many valuable ecosystem services and has been demonstrated to be a critical and essential habitat to many commercial and recreationally important species. The species abundance and diversity data from this study suggest that the shellfish aquaculture gear has similar habitat value for its inhabitants when compared with eelgrass. The species evenness data clearly shows that whereas the abundances may be greater in the SAG habitat, the SAG habitat is dominated by a few species.

The SAG habitat may also act as a predator refuge during early life stages of the lobster due to the limiting habitat within Point Judith Pond. In the spring and summer small lobsters are regularly found in the oyster cages and large predatory fish have been observed to frequent the aquaculture lease area including: the American shad (*Alosa sapidissima*), striped bass (*Morone saxatilis*), and winter flounder (*Pleuronectes americanus*). The American lobster supports an important fishery in the northeast United States therefore any habitat found to support the lobster should be considered commercially beneficial.

There is little research to date that describes the ecosystem services and benefits of aquaculture gear and its associated cultured product. The ecosystem services of the cultured bivalves and the benefits they provide to the marine ecosystem are fundamentally similar to those provided by wild stocks of bivalves. The aquaculture gear used to grow the cultured bivalves has intrinsic habitat complexity and shares many of the characteristics that artificial reefs possess. However, aquaculture gear is not a fixed structure, but it is periodically disturbed during maintenance and harvest operations. Most SAG habitat organisms are undoubtedly displaced during cleaning operations. Some of the sessile organisms are killed, but, the mobile species are probably able to quickly relocate to another of the 600 cages nearby when they are disturbed. The maintenance and cleaning of the aquaculture gear initiates recolonization of sessile invertebrate growth and inhabitance by motile organisms.

These findings indicate that shellfish aquaculture gear provides habitat for many native species of recreationally and commercially important fish and invertebrates in their early life history stages throughout the year. Therefore, we conclude that shellfish aquaculture gear has habitat value at least equal to and possibly superior to submerged aquatic vegetation. Future research should focus on growth, survival, and production of fish biomass within this habitat to further elucidate its habitat value.

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Aquaculture 290 (2009) 196–223

Contents lists available at ScienceDirect

Aquaculture

journal homepage: www.elsevier.com/locate/aqua-online

Review article

The ecological role of bivalve shellfish aquaculture in the estuarine environment: A review with application to oyster and clam culture in West Coast (USA) estuaries

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ARTICLE INFO

Article history:
Received 30 October 2008
Received in revised form 22 February 2009
Accepted 23 February 2009

Keywords:
Aquaculture
Disturbance
Eelgrass
Estuary
Bivalve
Ecology

ABSTRACT

Aquaculture is viewed as a potential mechanism to meet the growing demand for seafood around the world. The future of bivalve shellfish aquaculture in the U.S. hinges on sustainable practices on the part of industry and a more consistent regulatory regime. Bivalve shellfish aquaculture is a recent practice relative to its history in other countries, beginning in the late 1800s along the U.S. West Coast where it is now well established with farm raised product utilizing land-based hatcheries and grow-out directly in numerous estuaries. Bivalve shellfish aquaculture can be viewed as a disturbance which modifies the estuarine system in three ways: 1) changes in material processes — bivalves process food and produce wastes; 2) addition of physical structure — aquaculture introduces the cultured organisms and in some cases a physical anchoring structure; and 3) pulse disturbances like harvest and bed maintenance disturb sediments, remove species in addition to the cultured organisms themselves, and change resource or habitat availability. In U.S. West Coast estuaries, water column and sediment nutrient concentrations are relatively high and influenced by large tidal exchange and proximity to deeper nearshore ocean waters where upwelling controls production during summer months. Bivalves are unlikely to influence material processes except at local bed scales in these systems, although estuary-wide effects could appear as the fraction of cultured area rises or in poorly flushed bays. Bivalve culture clearly modifies estuarine habitat at local community and at landscape scales and effects are most often evaluated against existing structured habitat in the form of submerged aquatic vegetation. Individual activities act as pulse disturbances and the recovery of eelgrass (*Zostera marina*) to pre-disturbance levels is variable (<2 to >5 years). The extent of disturbance depends on the aquaculture practice and the distribution of eelgrass reflects a balance of space competition, pulse disturbance and recovery, and is therefore at dynamic equilibrium on aquaculture beds. Structure provided by aquaculture appears functionally similar to eelgrass for small benthic infauna and mobile epibenthic fauna while use of aquaculture as habitat by larger more mobile invertebrates and fish depends on mobility and varies with life-history stage and taxon being evaluated. Scale seems a very important management consideration and further research at estuarine landscape scales, especially for habitat use by important invertebrates and fish, may prove useful in designing and implementing best management practices. Though local and short term effects from aquaculture are clearly evident in U.S. West Coast estuaries, bivalve aquaculture does not remove area from the estuary or degrade water quality like other anthropogenic influences, and thus has not been implicated in shifts to alternate states or reduced adaptive capacity of the larger ecological system.

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0044-8486/$ – see front matter. Published by Elsevier B.V.
doi:10.1016/j.aquaculture.2009.02.033

1. Introduction

Aquaculture is increasingly viewed as a potential mechanism to meet the growing demand for food from the sea (Costa-Pierce, 2002), particularly as landings from world marine capture fisheries have plateaued (Brugere and Ridler, 2004; Muir, 2005). Although bivalve shellfish aquaculture represented only 10% of the world volume of fishery production in 2003, it represented 26% of world aquaculture production and 18% of world economic value (Lovatelli, 2006; Subasinghe, 2006). Bivalve shellfish production (both capture and aquaculture) has increased rapidly over the last 50 years from 1 million tonnes in 1950 to 13.2 million tonnes in 2003. However, 66% of that production is from China alone, whereas the U.S. produces less than 1% of the world's shellfish and leads the world in bivalve shellfish imports. Although domestic U.S. production is rising, concerns about environmental impacts currently constrain the U.S. industry, perhaps more substantially than in places like China, Japan and Southeast Asia where aquaculture has been a dominant and culturally accepted part of the coastal aesthetic for centuries (Kurokura, 2004; Costa-Pierce et al., 2005). The future of U.S. bivalve aquaculture hinges on sustainable mariculture practices on the part of industry and a more consistent regulatory regime, both of which were recently recognized as priorities by the U.S. Commission on Ocean Policy and the National Oceanic and Atmospheric Administration (USCOP, 2004; NOAA, 2006). Accomplishing these related goals will require an enhanced federal research program, however substantial research exists to help inform best practices and improve management decisions. In this review, we present an overview of the ecological issues associated with culture of clams and oysters in estuaries along the West Coast of North America.

Bivalve shellfish aquaculture in the U.S. is a recent practice relative to its history in other countries, beginning in the late 1800s with transfers of oysters (*Crassostrea virginica*) among East Coast estuaries and to West Coast estuaries as native populations were overfished (Baker, 1995; Lindsay and Simons, 1997; MacKenzie and Burrell, 1997). Today, farmed bivalves derive primarily from the West Coast; for instance, Washington state contributed 69% of U.S. production in 2002 (USDA, 2002 Census of Agriculture). However, the bulk of U.S. product still comes from capture fisheries along other coasts, where estuaries are larger, coastlines longer, and stocks of offshore clams newly accessible (Serchuk and Murawski, 1997). Contributions from these fisheries have declined recently, due in part to disease and overharvest in Chesapeake Bay, and a series of hurricanes along the U.S. Gulf Coast, yet the outlook for bivalve aquaculture is strong along all U.S. coastlines particularly given rising domestic demand and strong export markets in Asia (Harvey, 2006).

At the same time, bivalve shellfish aquaculture is experiencing increased regulatory scrutiny in the U.S. Human population density is rising along the coast, so more people are aware of aquaculture activities. Further, the cumulative effect of human activities now threatens estuarine habitat, water quality and native species. On the East Coast, much of the concern focuses on anthropogenic nutrient inputs, eutrophication, and other industrial pollutants (Kemp et al., 2005; Paerl et al., 2006; Smith et al., 2006). West Coast estuaries have been impacted by fecal coliform contamination and eutrophication in areas of Puget Sound, but the smaller outer coast estuaries have experienced greater change from introduced species and freshwater diversion and impoundment (Emmett et al., 2000; Kareiva et al., 2000; Borde et al., 2003; Thompson, 2005). While a number of U.S. federal, state, and local regulations address aquaculture activities, the most recent nexus for federal action comes from the U.S. Army Corps of Engineers (ACOE), which asserts jurisdiction under the Clean Water Act (Section 404) and the Rivers and Harbors Act (Section 10). In March 2007, the ACOE issued a new nationwide permit for shellfish aquaculture which in turn requires consultation with the US Fish and Wildlife Service (USFWS) and the National Oceanic and Atmospheric Administration (NOAA) for their regulatory authority under the Endangered Species Act (ESA) and Essential Fish Habitat (EFH) provisions of the Magnuson Stevens Fishery Conservation and Management Act.

Because the “best available science” with which these agencies have chosen to address regulatory issues is limited, especially for aquaculture on the West Coast, the agencies have tended to be very cautious about perceived impacts to habitats and/or communities of estuarine organisms that have been studied in greater detail elsewhere. Perhaps the best example is simply not permitting or requiring mitigation for aquaculture activities in areas where submerged aquatic vegetation is present, due to its recognized importance to fish and invertebrates elsewhere and a federal goal of “no net loss of wetlands” that is generally applied to any activity in these environments. This “precautionary” approach has directly affected existing aquaculture operations in California and Oregon, where growers have been forced to abandon historic culture areas or switch to off-bottom culture, particularly in areas where seagrasses are present (Chew, 2001; Rumrill and Poulton, 2004). A federal review is currently underway by ACOE and NOAA to determine how the laws discussed above will be administered on a nationwide basis. Commercial shellfish growers have taken a pro-active role in this issue by developing their own environmental management system, codes of practice and a regional research plan (PCSGA, 2001; PSI, 2005). This scientific review is a response to the need for establishing a baseline of relevant scientific information to inform impending management decisions.

In this review of the role of bivalve mariculture in estuarine ecosystems, we use an ecological framework that describes aquaculture practices as a disturbance (c.f. Simenstad and Fresh, 1995). We adopt the definition of disturbance used by Pickett and White (1985): “a disturbance is any relatively discrete event in time that disrupts ecosystem, community, or population structure and changes resources, substrate availability, or the physical environment”. In this context disturbances vary in their spatial scale, frequency, and intensity, which are relevant, respectively, to bed size, crop cycle, and type of aquaculture activity. We qualitatively distinguish "pulse" (short discrete events) from "press" disturbances (longer-lasting chronic events), despite their subjective definitions that imply a difference in the pace of the response (Glasby and Underwood, 1996). Nevertheless, the distinction is useful in considering, for instance, the effects of the farmed organisms, which are regularly present (press disturbance), and the effects of harvest operations, which occur periodically in the crop cycle (pulse disturbance). Disturbance has been viewed as a key process influencing marine benthic community dynamics for several decades (Sousa, 2001). Indeed, natural disturbances are essential components in the maintenance of community structure in some ecosystems (e.g. fire in many terrestrial ecosystems), even though they can have immediate negative effects on the abundance of some species. Anthropogenic disturbances may mimic such natural disturbances, although multiple, novel perturbations can exceed a system's capacity to maintain its characteristic state (Sousa, 1984; Paine et al., 1998). This capacity to recover, or resilience (Holling, 1973), depends on such factors as the extent of the disturbance relative to the mobility of key species, and the frequency of disturbance relative to generation time (Paine et al., 1998; Peterson et al., 1998). Recent ecological literature suggests that the likelihood of regime shifts to alternate states can increase when anthropogenic disturbance causes reduced "ecological" resilience (Gunderson, 2000; Folke et al., 2004; Scheffer et al., 2005; Groffman et al., 2006; Walker et al., 2006; Levin and Lubchenco, 2008). Using "disturbance" as a descriptor in a management context may involve a perception that disturbances are "negative". This is true only if the species of interest has "positive" value to the manager and the disturbance causes loss. Here, we describe aquaculture disturbance effects on species and ecosystems, but leave positive or negative value judgments to readers and managers.

Bivalve shellfish aquaculture influences the system in three primary ways: 1) material processes — bivalves process food and produce wastes; 2) physical structure — aquaculture introduces the cultured organisms themselves and in many cases a physical anchoring structure; and 3) pulse disturbances — harvest activities, in addition to some bed maintenance practices, can remove species in addition to the cultured organisms themselves, and change resource or habitat availability (Fig. 1). These influences occur on the time scale of the crop cycle (1–6 years, depending on area, method, and species) and do not include such longer-term changes as the introduction of non-native organisms during imports for aquaculture. Introduction of non-native species is an important management issue, particularly when they become invasive, but aquaculture and other vectors for marine invasions have been reviewed elsewhere (Gruet et al., 1976; Carlton and Mann, 1996; McKindsey et al., 2007; Minchin, 2007), and regulations and practices have changed to reduce the role of aquaculture imports in homogenizing biota (e.g. ICES Code of Practice on the Introductions and Transfers of Marine Organisms, ICES, 2005).

Our focus is on the intertidal culture of bivalve mollusks in estuaries along the West Coast of North America, particularly oysters because of the great spatial extent of estuarine area devoted to their culture and relatively large amount of research conducted to date. Although a similar review was conducted by Simenstad and Fresh (1995), substantial progress has been made toward quantifying the role of bivalve aquaculture in the estuarine environment since that time. Our intent is not to repeat their earlier effort, nor provide an exhaustive bibliography, but instead to update with new results and place in context with recent ecological literature from outside the U.S. West Coast. We begin with sections highlighting why these West Coast estuaries are distinct and how bivalves have been and are currently cultured there. Material processing, physical structure, and pulse disturbance associated with some aquaculture practices represent three interrelated aspects of the ecological role of shellfish aquaculture. Each topic is considered in its own section, first with generalizations from research worldwide, and then a summary of

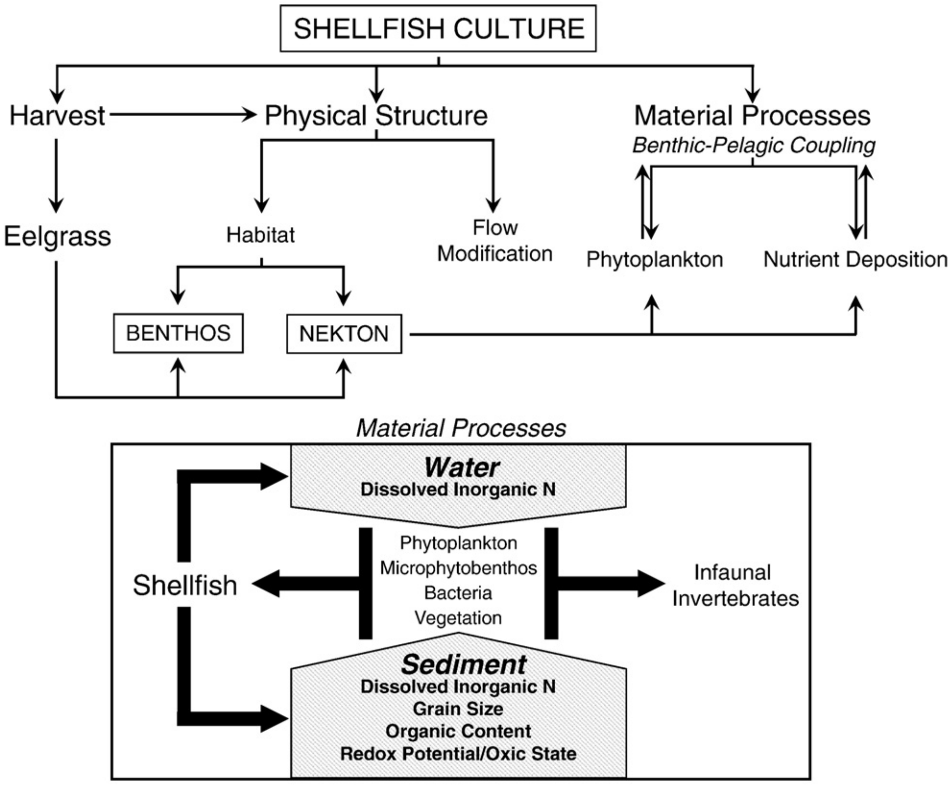

Fig. 1. Conceptual diagram of 3 pathways of shellfish aquaculture interaction with the environment. This simplistic view is of course more complex when broken down into component parts such as those for material processes including both benthic pelagic coupling and biodeposition (inset).

results from studies on the West Coast of North America in particular. In many cases, the evidence from West Coast estuaries aligns well with international perspectives; in some cases, the characteristics of West Coast estuaries lead to different ecological roles of shellfish aquaculture than have been demonstrated elsewhere; and finally, in most cases, the relatively nascent area of research on environmental impacts means that the picture for West Coast estuaries remains incomplete. In the final section, we address important research gaps, particularly at the landscape level that remain to be filled in order to make sound management decisions, though we recognize that such decisions clearly involve social and economic criteria as well.

2. Characteristics of West Coast estuaries

The major shellfish-growing areas on the West Coast of the U.S. include small coastal estuaries in California, Oregon, and Washington, as well as Puget Sound, which is a deep fjord extending several hundred km inland. Similarly, in British Columbia, Canada, bivalve aquaculture occurs both in small sounds on the west side of Vancouver Island and within the Strait of Georgia. Alaska's shellfish aquaculture industry is in its infancy but has grown markedly and oyster culture occurs primarily in remote geographic locations within areas like Kachemak Bay and Prince William Sound (Harrington, 2005; Oliveira et al., 2006). Four bivalve species contribute 99% of production from aquaculture in this region with Pacific oysters (*Crassostrea gigas*) representing 89%, and Washington leads production on a per-state basis, with production split about evenly between outer coast estuaries and Puget Sound (Table 1). West Coast estuaries have three general characteristics that set the context for aquaculture: they are geologically young and small; they have substantial oceanic influence; and they harbor distinct and relatively species-poor ecological communities. We treat each of these issues in turn.

Estuaries on the West Coast of North America are geologically young and relatively small compared to other well studied estuarine systems like Chesapeake Bay, but comparable in size to other shellfish-growing areas (Table 2). Only small remnants of the larger estuaries once present along the U.S. Pacific Coast in the Miocene and Pliocene epochs remain, due to subsequent uplift of the nearby landmass and estuarine infill (Jacobs et al., 2004). Further disturbances and oscillations due to glaciation and sea level change have subjected these estuaries to frequent flooding and emptying due to their small size (e.g. subsidences of 0.5 to 2 m have occurred multiple times with tectonic events in the last 7000 years (Atwater, 1987; Hagstrum et al., 2004). Combined with a relatively steep elevation gradient nearby, this causes these estuaries to provide much more spatially restricted and less stable environments than their larger counterparts on the Atlantic and Gulf Coasts. On the other hand, it would be incorrect to consider all of the region's estuaries as identical in abiotic features: substantial variation occurs in climate, mesoscale oceanographic features, and bathymetry, especially along the wide latitudinal range stretching from Alaska to California. Mesoscale oceanographic features establish faunal discontinuities (e.g. Cape Mendocino, Point Conception, Columbia River plume). Bathymetry varies because estuaries have formed at drowned river valleys (small coastal estuaries) and in deep fjords (especially Puget Sound and the Straight of Georgia; Emmett et al., 2000). Human impacts also vary regionally, for instance San Francisco estuary in California experienced substantial shoreline development, urbanization, and tideflat “reclamation”, which caused severe water quality impairment and abandonment by the oyster industry in 1939 (Barrett, 1963) as have other smaller estuaries in Southern California (Pitman, 1995). Smaller coastal estuaries of Northern California, Washington and Oregon generally have lower human population densities than does the Puget Sound trough, where water quality problems have also been more pervasive (Glasoe and Christy, 2004).

West Coast estuaries experience pronounced effects of the nearshore coastal ocean (Hickey and Banas, 2003). Coastal estuaries lie along an upwelling coast, and mesotidal conditions, combined with small size and extensive intertidal flats, contribute to substantial tidal exchange with the ocean (Table 2). For instance, Willapa Bay, Washington can exchange nearly half its volume during a single spring tide and the tidal exchange is about 40% in Coos Bay, Oregon. Even Puget Sound, due to the influx of ocean water at depth, shows signs (salinity, carbon) of ocean influence throughout its length (Babson et al., 2006; Ruesink et al., in prep). In addition, freshwater input is restricted during summer due to a Mediterranean climate pattern of winter rain and summer drought to the extent that smaller estuaries in California are hypersaline lagoons (Largier et al., 1997). Thus terrestrial inputs are generally less important for food webs than in other estuaries, because winter inputs coincide with low temperatures that limit productivity. Nevertheless local effects of rivers and small streams have been demonstrated in the riverine region of both Willapa Bay and smaller coastal estuaries in summer (Ruesink et al., 2003; Rumrill and Sowers, 2008). Winter inputs have been studied in San Francisco Bay where the North Bay shows evidence of terrestrially driven production despite anthropogenic water diversion (Thompson, 2005). Overall, oceanic conditions greatly influence both primary and secondary production within these systems (Roegner et al., 2002; Ruesink et al., 2003). Although the Mediterranean climate pattern ameliorates two types of natural disturbances with strong effects elsewhere in the U.S (winter ice in Northeast estuaries and hurricanes on the Southeast and Gulf Coasts), shallow intertidal flats in West Coast estuaries and extreme tidal exchange may nevertheless result in severe pulse disturbances from winter storms.

Although the continental shelf fauna along the West Coast is diverse, estuaries are relatively species-poor in a number of functional groups, possibly as a result of their comparatively short geologic history. We suspect that this short list of native species contributes to: 1) the availability of few native bivalves suitable for aquaculture, and therefore the adoption of primarily non-native species, 2) small numbers of estuarine-dependent fishes, and 3) “empty niches” for high-impact invasions. Cultured species in West Coast estuaries are predominantly non-native and have developed self-sustaining feral populations in some areas. This dependence on non-native bivalve species makes the U.S. West Coast similar to Europe (where Pacific oysters and Manila clams are also introduced and cultured), and different from the rest

Table 1
Yields (thousand pounds) by bivalve species from the West Coast of North America aquaculture in 2005 (from PCSGA, 2006; BC Ministry of Environment, 2006).

	Alaska	British Columbia	Washington	Oregon	California
Major growing areas	Kachemak Bay, Prince William Sound, Southeast Coast	Baynes Sound, Sunshine Coast, Cortez Isl., W. Vancouver	Willapa Bay, Grays Harbor, North Puget Sound, south Puget Sound	Coos Bay, Tillamook Bay	Humboldt Bay, Tomales Bay, Drakes Estero
Crassostrea gigas (Pacific oyster)	920	17,638	77,000	6290	10,000
Ruditapes philippinarum (Manila clam)	41	4188	8500		14
Mytilus galloprovincialis (Mediterranean mussel)	3		2100		600
Panopea abrupta (geoduck clam)			850		

Table 2

Comparison of system characteristics of several estuaries in Europe where aquaculture has been studied in detail, Chesapeake Bay, and U.S. West Coast estuaries.

Estuary	Type and ave. depth	Physical conditions	Residence time	Development and anthropogenic disturbance	Bivalves	Area total/ intertidal	Biomass, aquaculture area	References
Thau Lagoon, France	Lagoon with two small openings 4.5 m	Low wind, small tidal amplitude <1 m	5 months	Two towns	Oysters on longlines (80%), mussels (20%)	75 km^2/ <1 km^2	12,000 t harvested, 14–20,000 t present, 15 km^2	De-Casabianca et al. (1997), Souchu et al. (2001), Gangnery et al. (2003), Mazouni (2004), Mesnage et al. (2007), Metzger et al. (2007)
Sacco di Gorro, Italy	Lagoon with two 900 m openings 1.5 m	FW flow regulated	1–25 days	Towns, flow highly managed with dredging	Clams, mussels	26 km^2/ 0 km	15,000 t clams 1000 t mussels harvested, 8 km^2	Viaroli et al. (2003), Melia and Gatto (2005), Nizzoli et al. (2006), Marinov et al. (2007)
Marennes Oleron, France	Bay w/ Charente and Gironde Rivers 5m	3 m tidal range, low river flow, wind important	<10 days	Town, riverine nutrient influence	Oysters on trestles	136 km^2/ 82 km^2	30,000 t harvested 100,000 t present, 32 km^2	Raillard et al. (1993), Bacher et al. (1998), Gouleau et al. (2000), Leguerrier et al. (2004)
Chesapeake Bay, VA and MD	Drowned river valley, 8 m	Very large rivers (e.g. Susquehana) small tidal amplitude (0.7 m)	22 days	Large cities and towns, dredging	Clams, native oyster fishery	9900 km^2	940,000 t oysters in their heyday	Newell (1988), Gerritsen et al. (1994), Cerco and Noel, 2007
Totten Inlet, Puget Sound, WA	Portion of fjord 10 to >100 m	6 m tidal range, limited FW input	10 to 11 days	Rural, many waterfront residences	Clams, oysters, mussels and geoducks	24.7 km^2/ 0.85 km^2	1136 t oysters, clams and mussels	Brooks (2000)
Willapa Bay, WA	Bay w/ relatively small rivers, 3.2 m	Low river flow, tides and wind most important, 1.9 m tidal range, Columbia River influence	6–54 days at upper end	Towns, diked tidelands	Oysters on bottom and longlines	358/ 227 km^2	1468 t harvest, 46 km^2	Hedgepeth and Obrebski (1981), Hickey and Banas (2003), Ruesink et al. (2006), Banas et al. (2007)
Coos Bay, OR	Bay w/ relatively small rivers, 4 m	Low river flow, tides and wind important, 1.7 m tidal range	10–40 days	Small cities, lumber mills, diked tidelands, dredging	Oysters on bottom and longlines	34 km2/ 3.8 km^2	17 t	Hickey and Banas (2003), Rumrill (2006)
Humboldt Bay, CA	Bay w/ very small creeks, almost lagoonal, 3.5 m	Very low river flow, tides and wind important,	5 months in North Bay	Small Cities, dredging	Oysters on longlines	67/21 km^2	454 t, 260 ha	Barnhart et al. (1992), Rumrill and Poulton, 2004
Baynes Sound, British Columbia	Coastal portion of fjord, w/ Courtenay River and small creeks entering small embayments	Vertical stratification, of water column due to freshwater input and protection from wind and surface mixing	2 months for bottom water	Small towns	Oysters on bottom and deepwater rafts, clams with netting, wild clam harvest	87 km^2	850 t clams, 2510 t oysters, 458 ha	Jamieson et al. (2001), Carswell et al. (2006)

of the U.S. and Asia (Ruesink et al., 2005). West Coast estuaries contain fewer species of estuarine-dependent fish and invertebrates, particularly those that use estuaries as nurseries, than their U.S. East and Gulf Coast counterparts and display latitudinal trends in both abundance and diversity (Pearcy and Myers, 1974; Horn and Allen, 1976; Haedrich, 1983; Monaco et al., 1992; Nelson and Monaco, 2000). This could simply be due to small estuary size, but also the relatively large proportion of intertidal area, lack of significant freshwater input and their short geologic history. Similar latitudinal differences between estuaries and less diverse fish communities in small shallow estuaries have also been found in European, Tasmanian, South American, Australian and South African estuaries (Potter and Hyndes, 1999; Edgar et al., 2000; Araujo and de Azevedo, 2001; Pihl et al., 2002; Harrison and Whitfield, 2006). The mid- to high intertidal flats of West Coast estuaries are typically unstructured. High marsh occurs only above mean higher high water in West Coast estuaries, and seagrass (native eelgrass, *Zostera marina*) occurs around mean lower low water (Borde et al., 2003; Thom et al., 2003). With the exception of some relatively steep gravel and cobble beaches in fjords from Puget Sound, Washington north to Alaska, nearly all intertidal aquaculture activities take place on low gradient mud and sandflats, habitats that naturally have little structure in the region, except where aquaculture overlaps with native eelgrass. However some unwanted species have also entered this zone, forming structure in what is essentially an “empty niche”. These include several species of cordgrass (*Spartina* spp.) which have prompted multi-million dollar control efforts and, from British Columbia to northern California, an introduced seagrass, *Zostera japonica* (Daehler and Strong, 1996; Feist and Simenstad, 2000; Bando, 2006; Ruesink et al., 2006; Rumrill, 2006). Although species-poor in general, West Coast estuaries harbor several species of Pacific salmon (*Oncorhynchus* spp.), anadromous fish that spend variable amounts of their early life history in estuaries. Due to their cultural importance and longstanding fisheries, salmon drive substantial management activity and many subpopulations of salmon are extinct or listed as threatened or endangered under the US Endangered Species Act. Although the declines are likely from a variety of causes extending from freshwater to the ocean (Kareiva et al., 2000; Ruckelshaus et al., 2002; Good et al., 2007), loss and/or substantial modification of estuarine habitat may not be compatible with salmon recovery.

3. Historical fisheries and current farming practices

The most widely cultured bivalves in the United States are oysters, clams and mussels. Because of the presence of shells in middens, it is clear that bivalves have been harvested from North American estuaries for thousands of years (Trigger, 1986; Cannon, 2000). Coincident with European colonialism, extensive harvest of native oysters (*Ostrea lurida*) on the West Coast began in the mid-1800s (Baker, 1995), slightly after similar activity (for eastern oysters, *C. virginica*) along the Atlantic and Gulf Coasts in the 17th and 18th centuries (Kirby, 2004). Initial harvest implements included sailing vessels, poled bateaux, and a variety of hand tools (e.g. tongs), followed by dredges. Natural beds were gradually depleted on both U.S. coasts, in part due to the lack of return of shell material for natural recruitment (Kirby, 2004; Ruesink et al., 2005). When the native Pacific Coast oyster declined due to overharvest in the late 1800's, eastern oysters (*C. virginica*) and later Pacific or Japanese oysters (*C. gigas*) were transplanted to estuaries along the U.S. West Coast (Collins, 1892; Townsend, 1896; Steele, 1964; Sayce, 1976; Lindsay and Simons, 1997; Robinson, 1997; Shaw, 1997). From at least 1928 until 1977 (except during WWII), “seed” oysters were shipped from Japan to the U.S. West Coast annually for transplant. Additionally, after Pacific oysters established naturally-reproducing populations, some local production was possible. Waters were warm enough for “natural” spawning and setting to take place in Pendrell Sound and Ladysmith Harbor in British Columbia, and Dabob Bay and Willapa Bay, Washington (Scholz et al., 1984; Quayle, 1988).

With the advent of hatchery technology in the early 1980's (Nosho and Chew, 1991), oyster aquaculture along the West Coast of the U.S. became a completely integrated farming operation (see Conte et al., 1994 for a detailed description). Hatcheries are now essential to oyster aquaculture in Alaska, Oregon, and California, and contribute substantially in Washington and British Columbia, although naturally-set oysters continue to be incorporated when they are available. Most production involves *C. gigas*, but several other oyster species are also cultivated: *C. virginica*, *C. sikamea*, *C. ariakensis*, *Ostrea edulis*, and the native oyster *O. lurida*. In hatcheries, adult broodstock are conditioned and induced to spawn. Larval oysters are fed cultured phytoplankton until they are competent to settle and attach to a substrate (Muller-Feuga, 2000). This substrate is either pieces of shell (cultch) or ground shell or sand (cultchless). When the oysters are moved into estuaries, they are grown utilizing a variety of methods including bottom culture, floating bags, rack and bag systems, long lines and trays. In bottom culture, cultch with attached oysters is placed directly on intertidal (generally <0.6 m MLLW) and shallow subtidal bottom where it is left until the oysters reach market size, usually in one to three years depending on location and temperature. In Willapa Bay, where oyster growth varies substantially by area, oysters may be seeded to one area, allowed to grow for a period of one to two years and then transplanted to a second area called a fattening bed for final growth. Beds may also be harrowed with implements fashioned after the English pasture harrow (Sayce and Larson, 1966) to bring oysters back to the surface and break up clusters. Oysters are harvested (or collected for re-laying) from bottom culture by hand or with mechanical or suction dredges. In long line culture, seeded cultch is strung on lines or ropes that are suspended from stakes or rails and harvest is usually by hand. Cultchless oysters are often grown in high-density polyethylene (HDPE) or polypropylene mesh bags placed on the bottom, suspended off the bottom on racks, or placed in floating bags attached to longlines (Conte et al., 1994). Although not a focus of this review, suspended culture is typically used in areas that are relatively deep such as Puget Sound and especially British Columbia, Canada: seeded cultch is strung on lines or cultchless oysters are placed in trays or lantern nets, and these are suspended from floats. After harvest, single oysters are generally destined for the half shell market, and oyster clusters are either separated into singles, or processed at a shucking plant where the meats are packaged in containers for sale.

Oyster aquaculture contends with several pest species, most of which are predators or competitors and controlled by hand removal (Buhle et al., 2005). However, one species deserves special attention because its control involves the application of a chemical pesticide. Burrowing shrimp (*Neotrypaea californiensis*) are native deposit-feeders that bioturbate sediments up to 1 m in depth. At high densities, they preclude on-bottom oyster culture because the oysters are smothered or sink due to disturbance of the sediment by the shrimp. Since the 1960s, shrimp have been removed from oyster beds through the application of carbaryl (Sevin™), a general arthropocide (Feldman et al., 2000). Its use is now only legal in Willapa Bay and Grays Harbor, Washington where about 170 ha and 60 ha respectively (4% of the cultivated area and less than 1% of the tideflat in Willapa Bay) are treated by aerial application from a helicopter each year. Many species in addition to shrimp are killed by the pesticide, but the longer-term changes, including appearance of eelgrass, derive from the removal of shrimp and addition of oysters (Dumbauld et al., 2001; Dumbauld and Wyllie-Echeverria, 2003; Dumbauld et al., 2004). This pesticide is scheduled to be phased out of use in these estuaries by 2012, but the most effective alternative method of shrimp control discovered to date may also be chemical.

The West Coast harbors several native clams of historical importance to subsistence and commercial fisheries. These include: butter clams (*Saxidomus giganteus*), littlenecks (*Protothaca staminea*), horse clams or gapers (*Tresus capax* and *T. nutalli*), and geoducks (*Panopea abrupta*). On outer coast sandy beaches, razor clams (*Siliqua patula*) are fished commercially and recreationally (Lindsay and Simons, 1997). Eastern softshell clams (*Mya arenaria*) provided a small commercial fishery after their introduction from the East Coast in the late 1800's, but only during an initial post-invasion “boom” (Palacios et al., 2000) and more recently in Puget Sound. Clam aquaculture currently focuses on two species, one introduced (*Ruditapes philippinarum*, Manila clam) and one native (*P. abrupta*). Manila clams apparently arrived from Asia in the 1930s as a hitchhiker with oyster seed (Quayle, 1941; Chew, 1990). Growth in production occurred when growers were able to “seed” areas with small clams produced in hatcheries, at about the same time (1970s–80s) that such techniques were developed for oysters. Farmed Manila clams produced in hatcheries are held in trays or upwellers during a nursery period before being planted in a growout area or placed in mesh bags for growout (Toba et al., 1992). Several techniques are employed to enhance ground for clam production. Growers sometimes add gravel or oyster shell (Toba et al., 1992; Thompson, 1995), which provides substrate for the attachment of naturally-settled clams and likely makes feeding more difficult by some predators. Plastic or nylon netting of varying mesh is also often placed over clam beds to reduce predation. Manila clam aquaculture tends to occur higher (0.6–1.2 m MLLW) in the intertidal than does oyster culture. Harvest of planted tideflats is generally with a hand operated rake to collect clams, which grow close to the surface, but some mechanized harvest methods have also been developed (B. Dewey, pers. comm.). Worldwide, Manila clams are grown in China and Korea where they are native (Zhang and Yan, 2006), and in Europe from Italy to Great Britain, where they are introduced (Chew, 1990; Spencer et al., 1997).

A second clam species, the geoduck (*P. abrupta*) has recently become important for aquaculture in Washington and British Columbia. Geoducks are very large clams (up to 3.25 kg whole weight, Goodwin and Pease, 1991; Hoffmann et al., 2000) that have provided a valuable subtidal fishery for several decades. As a fishery, clams are not planted, and “recovery” of harvest tracts depends on recruitment which has only recently been assessed over appropriate temporal and spatial scales (Orensanz et al., 2004). Aquaculture techniques have been applied primarily to intertidal flats and have achieved crop cycles of ca. 5–6 years since growth is fast during initial years and then slows (clams can grow to be well over 100 years old; Goodwin, 1976; Shaul and Goodwin, 1982; Hoffmann et al., 2000). Geoduck culture techniques continue to evolve with survival in the hatchery and growout phases being highly variable. Protecting the hatchery produced geoduck “seed” from predation and drying out at low tide is essential. To date growers have largely used nursery tubes made by cutting 4–6 inch diameter PVC pipe into foot long sections and partially embedding them in the sediment. Several small (1 cm) geoducks are added, and mesh is placed over the top to exclude crabs and predatory snails (Beattie, 1992). This mesh may cover tubes individually or extend over an array of many tubes, anchored only at the edges (more like anti-predator nets used for Manila clams). The tubes are removed after 1–2 years, at which point the geoducks have reached a size and depth that avoids most predators and precludes desiccation at low tide. The geoducks continue to grow for several more years before reaching market size (15 cm shell length, approx. 1 kg whole weight). Harvest methods have been borrowed from the capture fishery and involve loosening the sediment around each geoduck with low pressure but high volume seawater forced through narrow tubes (stingers); geoducks are then removed by hand.

4. Shellfish as filter feeders and material processors

When abundant, suspension feeding bivalve mollusks can serve as important links between benthic and pelagic processes (benthic pelagic coupling) because they filter large volumes of suspended particles from the water column and eject them as both uningested pseudofeces and unassimilated feces which sink to the bottom (Newell, 2004). Bivalve aquaculture differs importantly from the culture of most finfish and crustaceans (Pohle et al., 2001; Crawford et al., 2003) in that cultured bivalves exploit naturally occurring phytoplankton at the base of the estuarine food chain, thus obviating the need for external feed inputs. For this reason, shellfish aquaculture does not result in additional nutrient loading, but rather, a transfer of nutrients from water column particles to benthic sediments in biodeposits, rapid nutrient cycling when dissolved inorganic nutrients are released into the overlying water, and a net removal of a portion of those nutrients when shellfish are harvested.

Particles filtered by bivalves range in size from bacterioplankton to less mobile zooplankton and include both living and non-living material, but most species are generally efficient at retaining material down to 3–5 μm (LeGall et al., 1997; Ward and Shumway, 2004; Prins and Escaravage, 2005; Lehane and Davenport, 2006; Lonsdale et al., 2007; Trottet et al., 2007). Bivalves can influence the community of plankton present via selective feeding favoring removal of picoplankton and phototrophic nanoflagellates over heterotrophic flagellates and diatoms in summer months at some locations (Bougrier et al., 1997; Wetz et al., 2002) but also feed on re-suspended microphytobenthos (Malet et al., 2007). Bivalves actively pump water through the gills and the particles are sorted on the ctenidia and labial palps. Capture efficiency increases non-linearly with particle diameter (Ward et al., 1998a). Captured particles are actively sorted by both size and quality (though the mechanism for the latter is not well studied; Shumway et al., 1985), and rejected particles become pseudofeces (Beninger et al., 1999). Oysters transport smaller phytoplankton to the dorsal ctenidial tract and detritus and larger particles to the ventral groove for sorting by the labial palps (Cognie et al., 2003). In contrast, particle sorting by mussels occurs primarily at the labial palps (Ward et al., 1997, 1998b). Oysters also tend to be able to cope with high seston loads by augmenting pseudofeces production, whereas clams and scallops lower their clearance rates as particle concentrations increase (Newell and Langdon, 1996; Defossez and Hawkins, 1997; Hawkins et al., 1998; Chauvaud et al., 2000). Finally some post-ingestive particle selection occurs in the stomach as the particle slurry moves across ridged sorting areas and pouches to the digestive diverticula (Cognie et al., 2001; Brillant and MacDonald, 2002, 2003). Assimilation efficiency varies with food source and how susceptible the particles are to enzymatic breakdown (Langdon and Newell, 1996). Some of the absorbed nitrogen is excreted as urine (primarily as NH_4). Both feces and pseudofeces are excreted by bivalves as mucous-bound aggregates with higher sinking velocity (up to 40×) than the ingested particles (Widdows et al., 1998). The particles gradually undergo a de-watering process and are incorporated into the sediment adding to the nitrogen and phosphorous pools (Deslous-Paoli et al., 1992; Hatcher et al., 1994). The feeding activities described here potentially allow bivalve shellfish aquaculture to alter larger material processes in the estuary in the following ways: water quality, sediment properties, and resources for primary producers indirectly through the release of inorganic nutrients. We treat each of these in turn.

4.1. Alteration of water quality

Measurable effects on water properties hinge on the filtration capacity of bivalves relative to the residence time of water in the estuary (Prins et al., 1998; Prins and Escaravage, 2005) — simplistically, longer residence times give suspension-feeders more opportunity to remove particles. This relationship is complicated however, not only by estuarine hydrography, but also because phytoplankton population growth, not just grazing, influences density, particularly if phytoplankton are supplied with readily available nutrients released by the grazers themselves or by anthropogenic sources (Dame, 1996; Prins et al., 1998; Chapelle et al., 2000; Souchu et al., 2001; Mazouni, 2004; Asmus and Asmus, 2005) Also, clearance rates are difficult to estimate in the field (Riisgard, 2001), sensitive to seasonal variation, and only relevant to the volume of water accessible to the benthos which can be modulated by structure created by the bivalves themselves (Lenihan et al., 1996). The contribution of cultured bivalves to clearance is further obscured when they represent an unknown fraction of all suspension-feeders (Heip et al., 1995; Leguerrier et al., 2004; Pomeroy et al., 2006; Dubois et al., 2007; Putland and Iverson, 2007), though this has also been measured and models developed to quantify it in some places (Sauriau et al., 1989; Leguerrier et al., 2004; Grizzle et al., 2008; Sequeira et al., 2008). Although it represents a relatively small contribution (1% of the meat weight), nitrogen is also removed from the system when actively growing shellfish are harvested (Lindahl et al. 2005). Finally, even if one is able to assess and detect alterations in water quality due to the presence of farmed bivalves, questions regarding scale and relevance such as whether they “improve water quality” or “exceed the system's carrying capacity” remain. We suggest that water clarity improvement will be more important in areas experiencing cultural eutrophication, and carrying capacity concerns will be informed by considering current relative to historic bivalve and other filter feeder densities.

Substantial work has been carried out worldwide on bivalve carrying capacity, especially via coupled biological–physical models (Dame and Prins, 1998; Gangnery et al., 2001; Sara and Mazzola, 2004; Duarte et al., 2005; McKindsey et al., 2006; Ferreira et al., 2007; Grant et al. 2008; Wang et al., 2008) because experimental work at the scale of whole estuaries is daunting. Several reports exist of bivalve growth rates declining as aquaculture densities increase over time, consistent with reduced phytoplankton densities available as food, particularly in areas such as lagoons with long water residence periods and/or shallow water column and high bivalve density often in three-dimensional culture systems (Rodhouse and Roden, 1987; Heral, 1993; Comeau et al., 2008). One recent monitoring effort documented the effects of removal of extensive oyster aquaculture from a eutrophic bay in Taiwan with mean chlorophyll levels and phytoplankton production being enhanced 4-fold in a less flushed portion of the bay (Huang et al., 2008). Invasive bivalves such as *Potamocorbula amurensis* in San Francisco Bay (Thompson, 2005), *Corbicula fluminea* in the Potomac River (Cohen et al., 1984), and dreissenids in the Hudson River and Great Lakes (Strayer et al., 1999) also provide evidence of top-down control of phytoplankton. Finally, depletion has been documented at spatial scales of individual aquaculture operations, and sophisticated models have been developed to explore the effects of hydrography and bivalve configuration on seston depletion at this scale (Newell et al., 1998; Drapeau et al., 2006; Grant et al., 2007). The results of these models align well with actual measurements of particle concentrations, but they generally indicate little impact on water properties beyond the immediate “footprint” of the aquaculture operation (but see Grant et al. 2008).

Many aquaculture areas on the West Coast of North America remain relatively pristine, that is, they have experienced low levels of cultural eutrophication. Consequently, carrying capacity concerns may be more relevant than the ability of cultured organisms to ameliorate water quality in these areas, although aquaculturists themselves can clearly be beneficial as an effective lobby for continued low-impact shoreline development (Steele, 1964; Glasoe and Christy, 2004). The calculation of present relative to historic bivalve densities has not been carried out for many locations, but in Willapa Bay, Washington, *C. gigas* occurs at >2.5 times historic biomass of *O. lurida* (Ruesink et al., 2005); the difference in filtration rate is probably less because the smaller native oyster should have higher mass-specific filtration. Willapa Bay shows a gradient in phytoplankton concentration from the mouth to upper estuary, and three competing hypotheses involve physical mixing of rich ocean and poor river water along the estuarine gradient, longer residence time of water in the upper estuary, and grazing by oysters that are farmed, especially near the mouth of the bay. Overlaying oyster filtration on a circulation model indicates that phytoplankton concentration declines into the bay more than would be expected from simple mixing, and the extra loss is consistent with the capacity of cultured oysters to filter it out (Banas et al., 2007). Interestingly, this result is achieved even though a large fraction of the bay's water (>80%) never moves over a shallow tideflat and is not susceptible to filtration (Banas et al., 2007). Empirically, however, the water that moves across Willapa Bay's tideflats is measurably affected by cultured oysters growing there. Wheat et al. (in prep.) documented declines of about 10%/100 m in phytoplankton when tracking parcels of water across oyster beds on flood tides (Fig. 2), whereas water flowing across newly-planted beds or other habitat types showed no

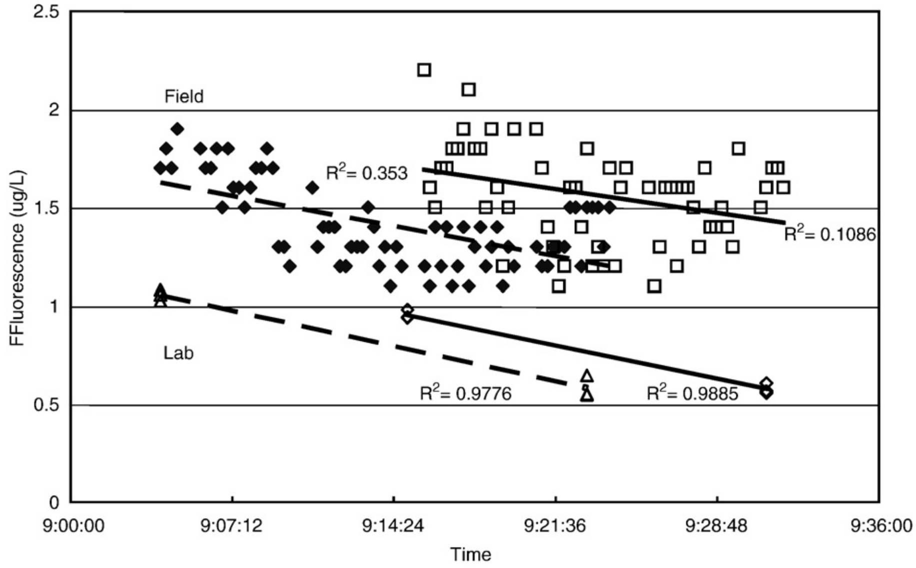

Fig. 2. Preliminary results from field surveys in Willapa Bay, Washington using drifting sensors (YSI® instruments) which moved over long-line oyster aquaculture areas and displayed a negative slope indicating phytoplankton (measured as chlorophyll *a* fluorescence) was being removed from the water column. Two separate drifts with instruments in the field are shown (top) and separate instrument calibration measurements taken with water bottles at start and end of drifts and measured on laboratory fluorometer (bottom).

such change. One West Coast area where public concern has been raised about aquaculture exceeding carrying capacity is in the southern end of Puget Sound, Washington. Interestingly, this is also one area where excessive nutrients from upland sources and some eutrophication have also been documented. However, the area of most extensive aquaculture (Totten Inlet) maintains higher bivalve growth than elsewhere in Puget Sound (Ruesink et al., in prep.), and only local phytoplankton depletion around raft structures has been documented (Cheney, pers. comm). Overall, existing evidence from the West Coast confirms that cultured bivalves affect water properties, but the effect is largely evident at small spatial scales (except see Banas et al., 2007). While water quality is certainly impaired in some West Coast shellfish-growing areas, this is most often due to the presence of fecal coliforms which do not affect water clarity and eutrophication has not yet been described as a significant problem in most areas.

4.2. Alteration of sediment properties

As with water quality, measurable effects of bivalves on sediment properties hinge on the density of shellfish relative to water flow. In the case of sediment alteration, however, the effects of bivalves are expected to be quite local, and flow enters into consideration because of its role in dispersing biodeposits (Callier et al., 2006). Much of the work on sediment effects has been carried out in the context of suspended culture. The buildup of biodeposits under or within culture operations reduces grain size and increases organic content, which can reduce oxygen content (Rodhouse and Roden, 1987; Christensen et al., 2003; Nizzoli et al., 2006; Mesnage et al., 2007; Richard et al., 2007; Hargrave et al., 2008). Nitrogen content in sediments (porewater) is often also elevated (Kautsky and Evans, 1987; Deslous-Paoli et al., 1992; Hatcher et al., 1994; Giles and Pilditch, 2006), but the more important consequence is for nitrogen cycling. Rates of NH_4^+ flux from bivalve communities can be fairly high (1–5 mmol N m^{-2} h^{-1}, Dame et al., 1992; Giles and Pilditch, 2006), however burial of N and P and removal of N from the system via denitrification (Fig. 1) are also enhanced by deposition from bivalves (Kaspar et al., 1985; Kautsky and Evans, 1987; Hatcher et al., 1994). Denitrification requires adjacent layers of oxygenated sediments supporting nitrifying bacteria and anaerobic conditions that support denitrifying anaerobes (Newell et al., 2002; Newell et al., 2005). Further feedback comes from microphytobenthos, which can intercept dissolved organic nitrogen being regenerated from the sediments and produce oxygen which further alters sediment chemistry and/or can be used by bacteria at the sediment–water interface to maintain nitrification (Rysgaard et al., 1995). These biochemical pathways are challenging to measure and quantify at larger scales in the field, for instance due to the presence of other nutrient regenerators like nekton in salt marsh creeks (Haertel-Borer et al., 2004). Laboratory studies are hampered by inappropriate boundary layer flows which were shown to greatly influence microphytobenthos and thus nutrient regeneration in mesocosm studies (Porter et al., 2004). A difference in nutrient cycling has also been observed across species. For instance on-bottom farming of clams stimulates transfer of both organic matter and oxygen to the sediment due to bioturbation by these animals whereas suspended mussel culture results in only the former and thus reduced oxygen (Nizzoli et al., 2006). Thus clam farming results in more balanced benthic metabolism with net losses of nitrogen from the sediment. The multitude of factors and number of ecosystem processes influenced by shellfish and complexity of the interactions make modeling efforts attractive tools for examining ecosystem effects on nutrient cycling (Cranford et al., 2007).

In general, there do not seem to be any characteristics of West Coast estuaries that would cause expectations of sediment effects to differ from those reported elsewhere, however sediment chemistry and chemical fluxes are largely unexplored there. When oyster aquaculture and natural habitat types were surveyed in Willapa Bay, the metabolic activity of aerobic microbes was lower in on-bottom oyster aquaculture, consistent with a larger contribution from anaerobes (Richardson et al., 2008), however no differences were found in sediment grain size, organic content, or oxidation–reduction potential. While these results may initially suggest weaker effects of cultured bivalves on sediment properties than have been observed in other systems, it is more likely that their effects were obscured in this comparative study by underlying sediment heterogeneity or the influence of local hydrodynamics. Indeed Rumrill and Poulton (2004) measured small-scale topographical alteration of sediment profiles directly beneath suspended long-line cultures of *C. gigas* in Humboldt Bay, California, and changes in sediment dynamics were most evident in experimental plots with high densities of oysters. Similarly experimental manipulation of oysters in Willapa Bay has produced sediment changes more consistent with those measured elsewhere: the addition of high-density oysters to small (2 m×2 m) plots in eelgrass reduced grain size and increased organic content relative to nearby controls (E. L. Wagner, unpubl. data). There was also evidence that oysters actually lowered ammonium concentrations in sediment porewater with one possible mechanism being rapid denitrification that occurs at the interface of organic and inorganic sediment layers created by biodeposition. In contrast, a similar experiment adding geoduck clams to small plots with eelgrass in Puget Sound resulted in slightly raised porewater ammonium concentrations, but grain size and organic content were apparently affected primarily by eelgrass and not the clams (Ruesink and Rowell, in prep).

4.3. Feedbacks to primary producers

The previous sections have addressed how bivalve feeding removes particles from the water and releases nutrients in two forms: packaged in biodeposits or dissolved into the water column and sediment porewater. These dissolved nutrients may stimulate the population growth of phytoplankton and benthic microphytobenthos (Kaspar et al., 1985; Swanberg, 1991; Mazouni, 2004; Asmus and Asmus, 2005; Sara, 2007). Microphytobenthos may be directly fertilized via biodeposition whereas phytoplankton may be either directly fertilized via ammonia release into the water or indirectly fertilized via re-suspension of biodeposits and/or stimulated nutrient recycling in the sediments (Fig. 1). Submerged aquatic vegetation and microphytobenthos may also benefit from improved light penetration if bivalves exert top-down control on phytoplankton (Phelps, 1994; Newell and Koch, 2004; Wall et al., 2008). Nutrients released by bivalves into sediment porewater can stimulate production of seagrasses, as documented for mussels growing with seagrass in Europe, Florida, and southern California (Reusch et al., 1994; Reusch and Williams, 1998; Peterson and Heck, 1999, 2001). In general an increase in sediment porewater ammonium toward optimal levels (~100 μM) should enhance plant biomass and growth (Dennison et al., 1987; Hughes et al., 2004), although high levels of biodeposits could lead to toxic sulfide concentrations especially in already eutrophic areas (Sorokin et al., 1999; Stenton-Dozey et al., 2001; Holmer et al., 2005; Vinther and Holmer, 2008; Vinther et al., 2008). Aquaculture also has the potential to stimulate competitors with seagrass, for instance providing attachment sites for epiphytic macroalgae and enriching nutrients used by algae (De Casabianca et al., 1997; Thomsen and McGlathery, 2006; Vinther and Holmer, 2008; Vinther et al., 2008). Seagrasses tend to be negatively affected by both epiphytic algal growth and macroalgal blooms (Hauxwell et al., 2001; McGlathery, 2001; Hauxwell et al., 2003).

What evidence exists for West Coast bivalve aquaculture to affect primary producers through nutrient or light pathways? In studies unrelated to aquaculture, eelgrass shoot growth and depth distribution have been shown to be light limited in Pacific Northwest estuaries (Thom et al., 2008). The addition of fertilizer to sediments around eelgrass growing in Padilla Bay resulted in NH_4 in excess of 1000 μmol l^{-1} and higher eelgrass shoot growth (approximately 0.5 cm/shoot/d; Williams and Ruckelshaus, 1993). Thus, to the extent that bivalves improve light availability or augment nutrient concentrations in sediment porewater, aquaculture could improve conditions for eelgrass growth. However total production of eelgrass was lower in aquaculture than in nearby eelgrass beds in Willapa Bay, due to a combination of reduced shoot density and smaller shoot size (Tallis et al., in press). The interaction has been explored experimentally in two additions of bivalves to eelgrass. On a per-shoot basis, eelgrass with geoducks grew faster than without (due to larger shoot size). The mechanism could be from a 20% enhancement of porewater ammonium, or from reduced intraspecific competition because eelgrass occurred at lower density in the presence of clams (Ruesink and Rowell, in prep.). In contrast, eelgrass with oysters grew slower than without (due to smaller shoot size). The plants may have been responding to an unexpected reduction in porewater ammonium associated with high-density oysters, but more likely were "clipped" by sharp shells (E.L. Wagner, unpubl. data). So far, the interaction between bivalves and eelgrass on the West Coast appears weak in terms of light or nutrient pathways (as opposed to strong effects via competition; see Section 5 below). Eelgrass growth responses to direct manipulations of light and nutrients have involved large changes in these drivers, whereas bivalves may not change the ambient levels so substantially. In any case, documented ambient porewater ammonium is close to the optimum for eelgrass growth in many of the region's estuaries, especially during summer months: Willapa Bay, Washington (80 µM, J.L. Ruesink, unpubl. data), Yaquina Bay, Oregon (60–170 µM and 750–2500 µM; Larned, 2003; Kaldy, 2006, respectively), San Diego Bay, California (20–120 µM; Reusch and Williams, 1998), south Puget Sound, Washington (50–90 µM; Ruesink and Rowell, in prep.), and Padilla Bay, Washington (30–137 µm; Williams and Ruckelshaus, 1993). Eelgrass growth appears consistently depressed below maximum when ammonium concentrations are below 40 µM (Dennison et al., 1987).

5. Press disturbance — shellfish aquaculture as structured habitat

Bivalve shellfish act as ecosystem engineers or foundation species by influencing habitat and resources available for other species (Jones et al., 1997; Bruno and Bertness, 2001). While engineering roles, such as the provision of complex hard substrate, are most obvious for oysters and mussels living above the substrate, all shell-producers including infaunal clams alter solute concentrations and their shells may contribute to surface structure after they perish (Palacios et al., 2000; Lehnert and Allen, 2002; Gutierrez et al., 2003). Ecosystem engineers have both positive and negative effects on ecological communities — they provide habitat and resources for some species, whereas other species may be displaced. Thus the effect of aquaculture involves both its "footprint" locally (for instance, percent physical cover within a small area) and regionally (for instance, density of farms), and its value for other species. Habitat "value" is not easily measured, as is evident in recent debates about "essential fish habitat" and "nursery habitat" (Able, 1999; Beck et al., 2001). Ideally, one is interested in how such habitats influence production of other species, but in practice lower-level indicators are generally measured: reproduction and survival, distribution and abundance, or simply presence and absence (Able, 1999). In this section, we first address the press disturbance resulting from ecosystem engineering by cultured organisms themselves, followed by structures added as part of aquaculture methods (e.g. stakes, tubes, racks, and nets).

The paradigm for soft-sediment tideflat communities is that they are structured by predation, which keeps prey densities at such low levels that larval recruitment and competition are relatively unimportant (Posey, 1990; Olafsson et al., 1994; Lenihan and Micheli, 2001). However, larger bioengineering species are an exception to this rule including large bioturbators (burrowing shrimp in West Coast estuaries; Posey, 1986; Posey et al., 1991; Dumbauld et al., 2001) and structure forming species like bivalve shellfish and eelgrass. Competition between cultured bivalves and eelgrass has been studied more thoroughly on this coast than in other locations, perhaps because aquaculture is so important and the two habitat types often co-occur or are adjacent. The relevance of eelgrass is two-fold: first, seagrasses in general are declining worldwide (Orth et al., 2006), and second, they form structured habitats and have consequently served as a benchmark for habitat provided by shellfish (Jackson et al., 2001; Heck et al., 2003; Bostrom et al., 2006). Because both shellfish and eelgrass shoots occupy space, it is no surprise that competition occurs. The focus of research has been to document the mechanism and strength of this competition. In terms of mechanism, oysters may influence eelgrass through both their "footprint" and abrasion or drying of leaves when exposed at low tide (Simenstad and Fresh, 1995; Schreffler and Griffen, 2000; Boese et al., 2003). Repeated damage is a possible explanation for smaller shoot sizes on aquaculture beds relative to nearby eelgrass (Tallis et al., in press). Living within the sediment, clams are not expected to cause leaf damage but may nevertheless compete for space. Geoduck clams at aquaculture densities (10 m^{-2}) reduced eelgrass density by ~30% in south Puget Sound during summer months; this difference disappeared during winter when shoot densities naturally thinned in control plots (Ruesink and Rowell, in prep.).

The strength of competition between shellfish and eelgrass can usefully be explored by examining how each species performs by itself and in combination (Fig. 3A, see example below). Some combinations generate overyielding, in which total production is larger than expected from averaging the 2 species. Overyielding can occur because individuals perform better in combination with another species than on their own. Other combinations result in underyielding, in which total production is lower than expected from the average of 2 species, often a result of strong interspecific competition. Thus, a key issue has been to elucidate the relationship between cultured bivalve density and eelgrass — does eelgrass do better or worse than expected from the percentage of area transformed to hard substrate? As a thought experiment, imagine that light limitation in the eelgrass canopy sets up low shoot density, and oysters at low densities simply insert themselves into unoccupied space. This may be the case, for instance, in an experimental addition of oysters at 10–20% cover in Willapa Bay, where eelgrass densities were not distinguishable from controls (shoot density = 25–50 m^{-2}; B.R. Dumbauld, unpubl. data). This would constitute a case of overyielding, because shoot density was apparently not reduced by the amount of oyster cover. On the other hand, to the extent that oysters damage nearby eelgrass shoots, it is possible to imagine eelgrass declining by more than the percent cover of oysters. At a site in the South Slough estuary (Coos Bay, Oregon) experimental addition of low densities of oysters (ca. 13 shells m^{-2} or 10% cover) resulted in a decline in eelgrass cover relative to an adjacent control plot (Rumrill and Christy, 1996), particularly at a higher tidal elevation (from 10% cover at deployment to 3% cover after 75 days versus an increase from 8% cover at deployment to 10% cover measured on the control plot). In Tillamook Bay, Oregon eelgrass shoot survival was only 1% in 3 × 3 m plots where oyster clusters (4–50 individuals) and eelgrass were transplanted compared to 59% survival in adjacent plots where only eelgrass was transplanted (Schreffler and Griffen, 2000). Both eelgrass and oyster cover may vary over time as demonstrated two years after clumps of oysters were added to 2 × 2 m plots in Willapa Bay, when eelgrass entirely disappeared from plots that had >20% oyster cover (Fig. 3B; E.L. Wagner, unpubl. data). Interestingly, however, a year later these plots showed a simple 1:1 tradeoff between oysters and eelgrass (Fig. 3C). At a still larger scale, on cobble shores in British Columbia, density of eelgrass transplanted down-slope from oysters declined more than density of similar transplants to areas not below oysters (Kelly and Volpe, 2007). If this spatial relationship was indeed causal, the competitive effects of oysters extended for several meters beyond their immediate location. Based on available evidence, eelgrass can coexist with bivalves at low densities used in on-bottom aquaculture on soft sediments, but more research needs to be directed at best management practices that could raise the likelihood of overyielding.

Introduced Pacific oysters are cultivated in many of the same West Coast estuaries once occupied by extensive beds of native oysters, *O. lurida*, raising the possibility of negative competitive effects between these oyster species. Despite relaxed fishing pressure, *O. lurida* has mostly failed to recover since it was overexploited in the late 1800's. Native oysters persist at very low abundance at many West coast locations and many factors likely contribute to their scarcity; however, one unexpected consequence of presence of cultured *C. gigas* in the intertidal is the development of a "recruitment sink" (Trimble et al., in press). Given a standardized recruitment surface (a stack of 11 Pacific oyster shells), native oysters disproportionately recruited to reefs of *C. gigas*, rather than eelgrass or unstructured tideflat in Willapa Bay, Washington. This gregarious settlement was an advantageous strategy when beds of native oysters were primarily found subtidally, but the modern shell distribution has shifted to be more intertidal due to *C. gigas* culture. Since *O. lurida* is sensitive to desiccation and temperature extremes, showing <5% annual survival at tidal elevations emerging from the water just 2–10% of the time, compared to >20% survival when continually submerged (Trimble et al., in press), native oyster recruits to intertidal shell habitat provided by *C. gigas* cannot persist.

Numerous studies have been conducted on the role of bivalves as habitat for fish and invertebrates in both estuarine and marine systems (Zimmerman et al., 1989; Jones et al., 1997; Breitberg, 1999; Coen et al., 1999; Posey et al., 1999; Bruno and Bertness, 2001; Coen and Grizzle, 2007; see discussion in Section 5.3 below), but most have concentrated on natural assemblages where these animals are considered to be foundation species or ecological engineers rather than aquaculture settings. Mussel and oyster reefs modulate water flow and transport, allow attachment of algae and invertebrates, and provide cover and refugia from predation (Bahr and Lanier, 1981; Zimmerman et al., 1989; Tokeshi and Romero, 1995; Lenihan, 1999; Ragnarsson and Raffaelli, 1999; Gutierrez et al., 2003; Rodney and Paynter, 2006). With the exception of restoration activities however, oysters and mussels in aquaculture settings are not generally allowed to form reef structures, but instead are either suspended on structures or spread out on the substrate to grow as individuals or small clusters and intentionally kept from forming three-dimensional reefs to positively influence valve shape and growth for market. Thus the ecological role of cultured bivalves as habitat, particularly when non-native, needs to be studied separately and not inferred from studies of bivalve reefs.

5.1. Benthic infauna

When suspended culture occurs over soft sediments, organic enrichment via biodeposition can transform a diverse benthic community dominated by suspension feeders (bivalves, crustaceans, and some polychaetes) into one dominated by smaller opportunistic deposit feeders (usually polychaetes), a pattern that characterizes nutrient enrichment from a variety of anthropogenic sources (Pearson and

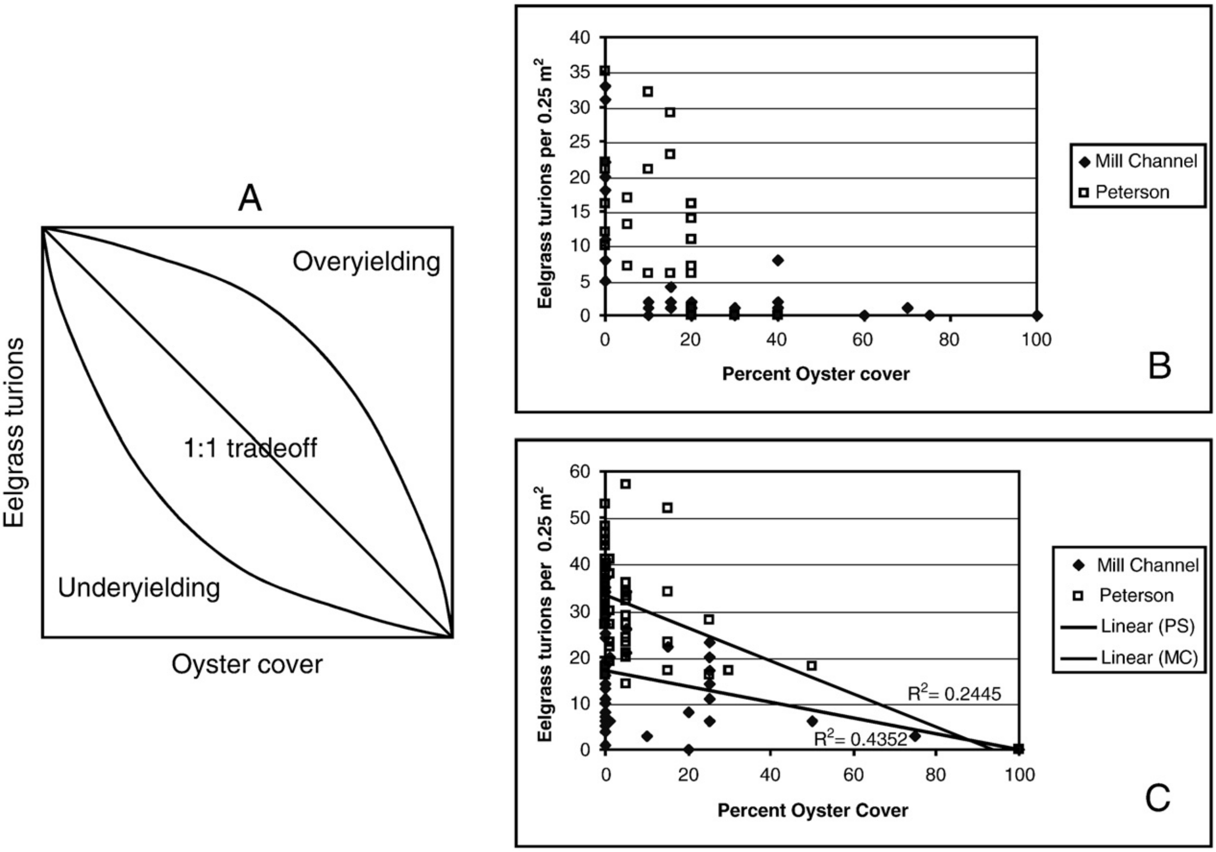

Fig. 3. A) Competition between eelgrass and oysters can be viewed on a continuum from overyielding, in which total production is larger than expected from averaging the 2 species because individuals perform better in combination, to underyielding, in which total production is lower than expected. B) Results of a field experiment in Willapa Bay, Washington where oysters were added to 2 m × 2 m plots at two locations (Mill Channel and Peterson Station). Measurements taken after 2 years indicated that eelgrass had entirely disappeared when oyster cover was greater than 20% (E.L. Wagner, unpubl. data). C) Measurements taken after 3 years showed some recovery and almost a 1:1 tradeoff between oysters and eelgrass.

Rosenberg, 1978). However, because biodeposition from shellfish farms is generally low relative to that of finfish farming or other anthropogenic sources (Pohle et al., 2001; Crawford et al., 2003), responses by infauna are variable and depend on species cultured, biomass or stocking density, and the environment in which culture takes place (Hartstein and Rowden, 2004; Miron et al., 2005; Callier et al., 2006). In some cases enrichment either does not occur or does not alter infauna (Grant et al., 1995; Chamberlain et al., 2001; Crawford et al., 2003), while in others, significant effects have been documented (Dahlback and Gunnarsson, 1981; Mattsson and Linden, 1983; Kaiser et al., 1998; Mirto et al., 2000; Christensen et al., 2003). Similarly, intertidal oyster culture on racks and trestles has variable consequences for infauna, likely based on the balance of biodeposit production versus water flow. Trestle culture of Pacific oysters in New Zealand, England, and France resulted in increased biodeposition, lower sediment redox potential, and altered macrofaunal communities (reduced diversity and abundance; Castel et al., 1989; Nugues et al., 1996; Forrest and Creese, 2006). However, oysters cultured in floating bags over intertidal areas in eastern Canada were shown to increase both organic content and initial colonization of de-faunated sediment by benthic infauna (Lu and Grant, 2008) whereas high currents at other sites in Canada and Ireland swept away biodeposits and few changes in infaunal community were seen (De Grave et al., 1998; Mallet et al., 2006). Finally, for on-bottom culture, it is difficult to separate effects of biodeposition from structural complexity and space competition and relatively few studies outside the West Coast of the U.S. have been conducted. Mussels cultured on the bottom were shown to negatively influence richness and abundance of some infauna in Wales (e.g. cirratulid polychaetes and two species of amphipods declined with increasing mussel density; Beadman et al., 2004) while Murray et al. (2007) found species dependent results when comparing communities in subtidal mussel culture (both on-bottom and rope) to naturally occurring intertidal mussel beds in Maine, U.S.

How do infaunal species respond to bivalve aquaculture on the West Coast? Several studies in West Coast estuaries have documented abundant, highly-diverse infaunal communities associated with on-bottom oyster culture. Abundance, biomass and diversity of infauna in Humboldt Bay on-bottom oyster culture were higher than that found in open mudflat, but lower than that in eelgrass (10.2 cm dia cores, 1 mm mesh; Trianni, 1995). At the time of this study oysters were harvested with a suction dredge in this estuary, a disturbance which likely also influenced the infaunal community in the oyster beds, and comparisons made suffered from statistical interaction between season and habitat factors. Similarly, Hosack et al. (2006) found infaunal macrofauna were more abundant in eelgrass in Willapa Bay, Washington than on open mudflat, and moderate levels found in oyster ground culture were not significantly different from either of these habitat types (10.5 cm diameter cores 0.5 mm mesh). Ferraro and Cole (2007) sampled multiple habitat types in Willapa Bay (2×8 cm diameter core, 0.5 mm mesh), using a strict sampling protocol throughout the estuary over two years. Oysters and eelgrass supported equally diverse assemblages, with unstructured habitats having less diversity. The equivalence of oyster and eelgrass habitats, in contrast to the other two studies, may have emerged from sampling aquaculture beds that had 2-3 year old oysters present and had not been disturbed for several years. The lowest-diversity samples came from areas influenced by burrowing shrimp (*N. californiensis*). This result is consistent with direct studies of bioturbation by these shrimp in U.S. West Coast estuaries (Bird, 1982; Posey, 1986; Dumbauld et al., 2001) and elsewhere around the world (Wynberg and Branch, 1994; Dittmann, 1996; Berkenbusch et al., 2000; Berkenbusch and Rowden, 2007; Pillay et al., 2007). Thus, the primary result of removing burrowing shrimp with the pesticide carbaryl is to reduce bioturbation and then add a three-dimensional architecture by planting oysters, which further influences the community (Dumbauld et al., 2001).

Other West Coast studies of infauna have occurred in aquaculture types that include structures (e.g. suspended bags, stakes, racks), so any differences reflect the presence of both bivalves and structures. Nevertheless, differences have been small. Although biodeposition was observed under deep-water suspended oyster and mussel culture using sediment traps in British Columbia, Canada, and Totten Inlet, Puget Sound, Washington respectively, there appeared to be little buildup of organic matter when measured outside the traps. The major change in the benthic community observed was enhanced abundance of epifaunal predators which capitalized on the drop-off of fouling organisms from the culture operations (Brooks, 2004; Switzer et al., 2008). Harbin-Ireland (2004) found no difference in percent organic matter in areas directly below and those adjacent to a single set of subtidal oyster racks in Drakes Estero, California, nor was there any difference in overall infaunal community indices. The relative abundance of amphipods was higher under racks while other taxa like bivalves and ostracods were less abundant. In contrast, experimental deployment of oyster stakes and racks in Coos Bay, Oregon, resulted in biodeposition within the stake plots and erosion and reduction of carbon content of the sediments below rack plots (Everett et al., 1995). Stake culture resulted in lower recruitment and survival of tellinid clams, while increased abundance of cumaceans and amphipods was observed in the oyster rack plots (Carlton et al., 1991). In a relatively short term follow-up study, Pregnall (1993) observed no significant difference in sediment grain size and only a slight difference in benthic infaunal diversity between oyster stake culture plots and eelgrass controls. Diversity and abundance of infaunal invertebrates around long line oyster culture in Humboldt Bay, California were also similar to those observed at eelgrass reference areas (Rumrill and Poulton, 2004). In both cases, similarity may have arisen not simply due to flow dispersing biodeposits, but because both aquaculture and control areas included eelgrass, which has characteristic effects on sediment (reducing flow, allowing particles to settle out; Madsen et al., 2001).

From the above discussion, it is clear that engineered habitats differ from unstructured tideflat in terms of static community-level metrics. Differences have also emerged from the few but important studies addressing species interactions. Small mobile crustaceans including juvenile Dungeness crab (*Cancer magister*) had higher density and biomass in oyster shell habitat placed in intertidal areas of Grays Harbor, Washington than in nearby control areas without shell. When small clams recruited to the structured shell habitat in higher numbers, they also experienced higher predation by crabs (Iribarne et al., 1995; Dumbauld et al., 2000).

5.2. Epibenthic meiofauna

Epibenthic invertebrates are important food items for juvenile estuarine fish (Alheit and Scheibel, 1982; Gee, 1989) including juvenile Pacific salmon and small (<50 mm) English sole on the U.S. West Coast (Toole, 1980; Simenstad et al., 1982). Structurally complex habitats like seagrass have been shown to enhance meiofaunal abundance (Bell et al., 1984; Attrill et al., 2000; Jenkins et al., 2002). We found only two studies on the effects of either oysters or aquaculture on epibenthic meiofauna. Castel et al. (1989) found enhanced epibenthic meiofaunal abundance in both on-bottom and bag culture of oysters in France over that found in adjacent un-structured habitat, but highest abundance was found in nearby seagrass beds. Similarly meiofauna densities were higher in both eelgrass and oyster habitats than over open mudflat in Willapa Bay on the West Coast (Hosack et al., 2006). Simenstad and Fresh (1995) noted that taxa diversity of epibenthic harpacticoid copepods was higher on an active on-bottom oyster culture plot with 3 year old oysters present than an in-active plot where oysters and eelgrass were present in the same estuary, but prey taxa for epibenthic feeding fish such as juvenile salmonids were more abundant on the in-active plot. This trend was reversed for English sole prey taxa suggesting species specific differences in affinity, but no comparisons were made with eelgrass or other habitats. Recent experimental work with experimental additions of the much smaller native oyster (*O. lurida*) attached to Pacific oyster cultch shells in Puget Sound, Washington also showed increased abundance of epibenthic organisms, but not necessarily enhanced taxa richness with the pool of available species being determined by background conditions at the enhancement site (Cordell, pers. comm.).

5.3. Nekton

For larger mobile species, complex structure formed by foundation species or ecosystem engineers is likely to provide a place both to search for prey and to avoid becoming prey to larger organisms. Indeed, higher densities of estuarine fish and invertebrates have been widely found in association with structured habitats like seagrass (Orth et al., 1984; Jackson et al., 2001; Heck et al., 2003) and oyster reefs (Breitberg, 1999; Coen et al., 1999; Lenihan et al., 2001; Lehnert and Allen, 2002; Glancy et al., 2003; Peterson et al., 2003; Coen and Grizzle, 2007; Horinouchi, 2007; Taylor and Bushek, 2008) when compared to open un-structured mudflat or subtidal channel bottom. However, abundance measures are not a definitive indicator of how structured habitat contributes to nekton production, and in some cases even densities do not differ across habitat types (Heck and Thoman, 1984; Ferrell and Bell, 1991; Jenkins et al., 1997). We discuss possible explanations for these discrepancies in Section 7 below. Nekton response to aquaculture as habitat has not been extensively studied, except in the context of off-bottom culture operations. Order of magnitude higher densities of some fish and invertebrates (American eel, oyster toadfish, rock gunnel, Atlantic tomcod, and American lobster) were found in areas with rack and bag culture of *C. virginica* in Rhode Island, USA, compared to those in eelgrass or unstructured habitats, but eelgrass also harbored a few unique species (northern pipefish and winter flounder; Dealteris et al., 2004). Tautog and scup were more abundant at oyster grow-out sites than natural rocky reefs in Naragansett Bay, Rhode Island and a tagging study found that though scup grew at slightly higher rates on the rocky reefs, they had greater site fidelity to oyster grow-out cages (Tallman and Forrester, 2007). Erbland and Ozbay (2008) found higher abundance of several reef oriented fish species (gag grouper, grey snapper, sheepshead, and tautog) and greater overall species richness in experimental oyster bags compared to a nearby oyster reef in Delaware. Juvenile sole were found to utilize areas with oyster trestle culture for protection during the day while foraging on surrounding tideflats at night (Laffargue et al., 2006). Researchers in New Zealand established a framework for the expected effect of suspended culture on fish which includes three mechanisms: attraction to structure, direct influence on recruitment, and indirect food web effects (Gibbs, 2004). A case study which examined suspended culture of green mussels suggested few realized effects on abundance of one species (blue cod) and that the primary effect might be on pelagic fish that consume zooplankton should the footprint of farms be expanded (Jiang and Gibbs, 2005). These mussel farms have also been shown to enhance abundance and aggregation of starfish on the bottom, presumably due to drop-off of both culture species and fouling organisms (Inglis and Gust, 2003 D'Amours et al., 2008). Clynick et al. (2008) found species specific differences in abundance when comparing areas under mussel culture lines to adjacent eelgrass and open unstructured habitat, but found no differences in integrated growth of winter flounder, sand shrimp and rock crab measured using RNA/DNA ratios.

Substantial research on nekton associated with both on-bottom and long-line oyster culture has been carried out recently along the West Coast of the U.S. In one case, diversity and abundance of fish were highest in aquaculture. Specifically, oyster longlines in Humboldt Bay, California, harbored more fish than did eelgrass or open mud habitats (Pinnix et al. 2005). However a more common result has been that community-level indices (abundance and diversity) are equivalent across habitats with a few species specific affiliations. Few statistically significant differences in density were found among the >20 species of fish and crabs collected at intertidal locations in Willapa Bay, Washington where eelgrass, oyster bottom culture and open mudflat habitats were surveyed (Dumbauld et al., 2005; Hosack et al., 2006, Fig. 4). In general, nekton density reflected physical location in the estuary rather than habitat type, although some species like rock crab (*Cancer productus*) were more abundant in oyster aquaculture and tube snouts (*Aulorhyncus flavidus*) in eelgrass. Higher abundance of rock crab and smaller shore crabs (*Hemigrapsus* spp.), sculpins and blennies, occurred in small oyster stake culture plots compared with nearby eelgrass control areas in Coos Bay, Oregon (Pregnall, 1993). Staghorn sculpin (*Leptocottus armatus*) and caridean shrimp (*Crangon franciscorum*) were more abundant within high-density oyster long-line plots compared to lower-density oyster plots in Humboldt Bay, California (Rumrill and Poulton, 2004). In a study of fish associated with oyster racks, no significant differences in species richness or abundance were observed in fish samples collected adjacent to the racks compared to an area without culture in separate arms of Drakes Estero, California. At the same time, structure-oriented feeders like kelp surfperch (*Brachyistius frenatus*) and crevice-dwelling fish like gunnels and kelpfish (*Pholis ornata* and *Gibbonsia metzi*) were disproportionately associated with racks particularly during the day when refuge from predators could be most important (Weschler, 2004). Although oysters *C. gigas* were not included in the comparison, adjacent cobble habitats, supported lower fish diversity than eelgrass in British Columbia (Kelly et al., 2008).

On-bottom structure appears to have different implications for Dungeness crab (*C. magister*), depending on phase of the life cycle. These crabs represent a multi-million dollar annual fishery on the U.S. West Coast and the role of estuaries as nurseries supporting these populations has been extensively studied (Armstrong et al., 2003). Ground cultured oysters and intertidal shell provide equal or better habitat than eelgrass for juvenile 0+ Dungeness crab (0–30 mm carapace width), which in turn provides better habitat than open

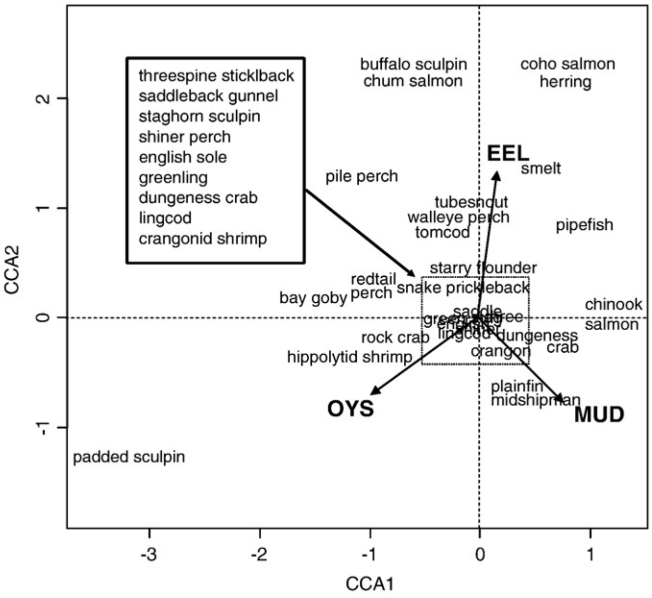

Fig. 4. Cannonical correlation biplot of catch per unit effort data from fyke nets deployed over three intertidal habitats: oyster aquaculture (OYS), eelgrass, *Zostera marina* (EEL), and open mud (MUD) in Willapa Bay, Washington in 2001. Species close to vectors and far from the midpoint are closely associated with that habitat. Many commonly collected species (box and circle) show no association with habitat, but some others like tubesnout and smelt in eelgrass, and rock crab and hippolytid shrimp in oyster show loose association with habitat.

unstructured mud or sand based on higher recruitment and survival rates due to protection from predators (Armstrong et al., 1994; Eggleston and Armstrong, 1995; Dumbauld et al., 2000; Feldman et al., 2000; Williamson, 2006). Shore crabs (*Hemigrapsus oregonsensis*) can also recruit to shell and outcompete *C. magister* depending on location and tidal elevation (Visser et al., 2004). Older age classes of Dungeness crab (1+ and 2+), however, favor open unstructured littoral habitats for foraging at high tide and are less likely to move across structured habitat including both eelgrass and oyster aquaculture (Holsman et al., 2003; Holsman et al., 2006).

For some Pacific salmon, on-bottom oysters appear to be lower quality habitat than eelgrass, especially for predator avoidance, based on individual behavior. However, salmon are distributed broadly across habitat types, and the amount of aquaculture in an estuary does not appear to influence salmon returns. Pacific salmon occupy estuaries during a critical life-history stage as juveniles smoltify and transition from fresh to marine waters (Quinn, 2004). The diversity of life-history patterns among and within species of salmon influences their use of estuaries: Chinook salmon (*Oncorhyncus tschawytscha*) have the longest estuarine residence, whereas pink and chum salmon (*O. gorbuscha* and *O. keta*) move through estuaries relatively rapidly on their way to sea (Groot and Margolis, 1991; Bottom et al., 2005). Juvenile coho salmon (*O. mykiss*) generally move directly to neritic waters as older 1+ fish that have reared in freshwater for longer periods, but recent evidence suggests that both age-0 and yearling fish utilize estuarine areas (mostly salt marsh and sloughs) relatively extensively (Healey, 1982; Miller and Simenstad, 1997; Miller and Sadro, 2003). Across West Coast estuaries, successful returns of salmon do not appear to be related to the presence or absence of aquaculture; in fact, the best estuarine predictor of pit-tag returns of Chinook salmon was the percent of land cover in natural condition (Magnusson and Hilborn, 2003). In a field study to assess habitat preferences, salmon smolts were sampled across habitat types in Willapa Bay, Washington with a towed net, and gut lavage performed on captured individuals. No differences in abundance or diet occurred across habitat types, although seasonal and larger-scale spatial differences were evident (Dumbauld et al. 2005, Fig. 5). Laboratory studies of Chinook salmon smolt behavior indicated that eelgrass may provide a better refuge than other habitat types. Larger juveniles preferred the structure of eelgass as a refuge over oysters or open sand substrate when exposed to a mock heron predator (Dumbauld et al., 2005). In a separate field experiment, hatchery-raised Chinook salmon smolts were released into a large intertidal enclosure (3000 m^2) containing eelgrass (*Z. marina* and *Z. japonica*), oyster clusters, unstructured sediment, and introduced cordgrass (*Spartina alterniflora*). They were implanted with acoustic tags that allowed their movements to be tracked in 2-dimensions at sub-meter accuracy (Radio Acoustic Positioning and Telemetry). After effects of tidal elevation and enclosure were accounted for, smolts never entered areas with *Spartina* and otherwise responded only to the presence of *Z. marina*, where they moved more slowly than in other habitat types (Semmens, 2008). Thus it seems that juvenile salmon move over the entire matrix of estuarine habitats, but eelgrass may represent a preferred habitat for refuge from predation which cannot be compensated by transformation to on-bottom oyster aquaculture. Assessing the functional value of habitats including aquaculture however, will ultimately require a larger landscape approach as suggested by Simenstad and Cordell (2000) for restoration (see further discussion in Section 7 below).

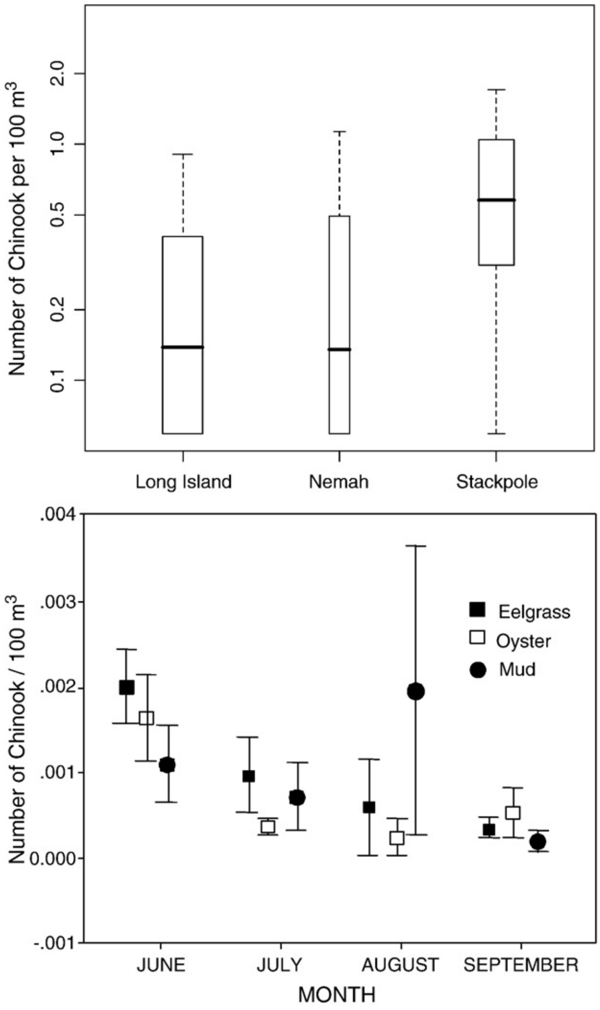

Fig. 5. Mean catch per unit water flow of juvenile Chinook salmon smolts by area fished with a modified two boat trawl net in Willapa Bay, Washington during 2003–2005. Catch varied by location (top), and over time (bottom), but no difference was evident between habitats.

5.4. Birds

Estuarine tidelands provide foraging habitat for numerous species of shorebirds, waders and waterfowl during migration and for a few species that overwinter. Some farmed bivalves are directly consumed by birds (e.g. mussels by seaducks and oystercatchers; Caldow et al., 2004). However, other bird species appear to avoid densely-structured habitats, preferring instead to feed in open mudflat areas (Luckenbach, 1984). Like nekton, the response of birds to aquaculture is likely to be species and perhaps environment specific due to bird feeding and roosting behavior relative to the tides and the presence of other birds and predators. In Ireland, dunlin (*Calidris* spp.) were more frequent beneath trestle cultured oysters, whereas gulls, curlew and oystercatchers occurred in significantly lower numbers in culture areas (Hilgerloh et al., 2001). Experimental additions of mussels to intertidal areas in Wales resulted in increased use by curlew and redshank over time (Caldow et al., 2003), due apparently to increased diversity in benthic fauna as food provided by increased habitat complexity.

Evidence for the effects of aquaculture on birds on the West Coast suggests species specific differences due to behavior. In Humboldt Bay, California five of 13 species of shorebirds (whimbrel, willet, dowitchers, peeps and black turnstones) and three of four species of waders (snowy egret, great egret, and black-crowned night heron) were more abundant in long-line oyster culture areas than in nearby “control” areas (Connolly and Colwell, 2005), possibly responding to higher densities of invertebrate prey associated with long-lines. Black bellied plovers and great blue herons were more abundant in control areas and the other shorebirds displayed location specific behavior. Kelly et al. (1996) found that peeps and dunlin avoided rack and bag oyster culture in Tomales Bay while willet were attracted. Mussels settling on aquaculture structures were shown to enhance food resources for surf scoters and Barrow's goldeneye in British Columbia (Kirk et al., 2007). One species of waterfowl of particular concern are Brant geese (*Branta bernicla*), which graze heavily on eelgrass in estuaries at stopovers during their long migration to Arctic breeding grounds and in temperate over-wintering areas (Ganter, 2000). Humboldt Bay, California ranks fourth among West Coast spring staging areas for brant with peak numbers reaching 38,000 while Willapa Bay historically averaged 23,393 (1936–1960; and now 6900) and ranks sixth (Wilson and Atkinson, 1995; Moore et al., 2004). Effects of geese themselves on eelgrass tend to be low due to the seasonally brief presence of these birds during migration, but Canada geese (*Branta canadensis*) as well as several species of dabbling ducks have been shown to have more substantial long term effects on eelgrass in areas elsewhere in the U.S., particularly where they overwinter in some years (Nacken and Reise, 2000; Rivers and Short, 2007). Dabbling ducks have switched their foraging habits to feed primarily on the introduced species of eelgrass *Z. japonica* in West Coast estuaries where this species is now present (Lovvorn and Baldwin, 1996). While goose density is positively correlated with spatial eelgrass coverage in West Coast estuaries, the relationship is more complex and also influenced by proximity to the next estuary along the coast and other factors like frequency of winter storms and within season foraging dynamics due to tides (Baldwin and Lovvorn, 1994; Wilson and Atkinson, 1995; Moore et al., 2004; Moore and Black, 2006), Thus both eelgrass and associated brant numbers could be negatively associated with aquaculture in a given area, but the long-term temporal decline in brant numbers along the West coast is not likely associated with shellfish aquaculture given the relatively stable presence of this industry in these systems for the last 100 years.

5.5. Aquaculture structures

Some bivalve aquaculture methods introduce physical structures to the intertidal flat in addition to the organisms themselves (e.g. stakes, longlines, and bags for off-bottom oyster culture, and gravel, tubes and anti-predator nets for clam culture). These physical structures can modify water flow, in some cases accelerating flow and causing erosion, in other cases leading to deposition. They can also provide attachment sites and attract settlement of other invertebrates and algae. For eelgrass in particular, reduced density is a common response to the shade from overwater structures, studied most intensively for docks (Burdick and Short, 1999; Thom et al., 2005). We have already considered effects of suspended and rack culture in sections above, although the species versus method effects were not distinguishable. Distinguishing these effects requires two treatments (species + method, and either species alone, or method alone [preferably both]), in addition to a control without either. This design was employed by Spencer et al. (1997), who showed that predator netting deployed to protect clam aquaculture substantially increased sedimentation to tideflats in Great Britain, whereas biodeposits from clams alone did not (but see Jie et al., 2001 who document increased biodeposits from clams can occur in areas with current velocities below critical re-suspension rates). Further, this sedimentation resulted in slightly enhanced organic content and enhanced abundance of deposit feeding polychaetes. Clam mariculture conducted in bags on the East Coast of the U.S. has been shown to affect sediment but not water column characteristics (Mojica and Nelson, 1993) Macro-algae attached to clam culture bags in North Carolina enhanced use by mobile invertebrates and juvenile fishes over that in nearby shallow subtidal sand bottom and resulted in comparable abundances with seagrass habitat (Powers et al., 2007). Both substrate modification (gravel addition) and predator netting effectively increase survival by protecting juvenile seed clams (*Mercenaria mercenaria* and *M. arenaria*) from various predators on the East Coast of the U.S. (Kraeuter and Castagna, 1985; Beal and Kraus, 2002) and *R. philippinarum* in Spain (Cigarria and Fernandez, 2000), but these studies were primarily confined to effects on the clams themselves.

What evidence exists for ecological effects of aquaculture structures on the West Coast? For oyster culture most regional attention has focused on response by eelgrass (*Z. marina*) and results have been quite variable. At one extreme, oyster stake culture conducted in the middle of an intertidal eelgrass meadow in Coos Bay, Oregon reduced eelgrass cover by 75% relative to nearby controls, possibly due to increased sedimentation (5–10 cm buildup) and physical disturbance (Everett et al., 1995). Oyster racks caused 100% loss of eelgrass under the structure from both erosion of sediment (10–15 cm around structure) and shading. Macro-algal biomass was enhanced around stakes and significantly lower in rack plots than in eelgrass reference plots (Everett et al., 1995). At the other extreme, a broad survey of Willapa Bay showed that eelgrass density in longlines could not be distinguished from uncultured areas at the same tidal elevation, although in a subset of these beds, longlines harbored smaller plants (32%) and reduced production per unit area (70%) (Tallis et al., in press). Also, in a separate study in Willapa Bay, lower eelgrass densities were found in longlines than in nearby eelgrass reference areas (Wisehart et al. 2007). Seedlings were less abundant in longlines and reference areas compared with dredge harvest beds, possibly from seed supply or because shading and sedimentation impact these small plants (Wisehart et al. 2007). In an experimental study in which the effect of space between oyster longlines on eelgrass was examined in Humboldt Bay, California, eelgrass metrics tended to scale directly with the density of oysters (Rumrill and Poulton, 2004). Eelgrass declined in cover and density as spacing between lines decreased; spatial cover and density of eelgrass under lines spaced at 1.5 ft and 2.5 ft were significantly lower than those spaced at 5 and 10 feet apart. Eelgrass metrics observed within these wider spaced lower-density treatments were comparable to those observed at a nearby untreated site, a former oyster ground culture site, and a series of eelgrass reference sites located throughout the bay at the end of the 22 month study period (Fig. 6). They were also comparable to those measured within full-scale commercial long-line culture areas. Increased sedimentation and more variable light conditions (incident light levels diminished by as much as 35%) were found under narrowly spaced long-lines (<5 foot spacing), but the "shade zone" migrated with movement of the sun and irradiance was not reduced enough to limit *Z. marina* growth. Structures clearly have the potential to limit eelgrass, but the effects are context specific, may not even be apparent when observed over larger spatial and longer temporal scales (e.g. see Ward et al. 1993 for lack of observed effects of rack culture), and can be ameliorated with management practices. West coast growers have also reported that eelgrass often appears in areas formerly devoid of this plant after structures are put in place. Given the lack of evidence for nutrient enhancement (Section 4.3 above), this could be due to either localized effects on water clarity or sediment stabilization, but no studies have addressed the mechanism.

For clam culture on the West Coast, two modifications have been studied: addition of shell or gravel to the substrate and addition of anti-predator nets. Gravel and crushed oyster shell have been widely used to develop or maintain hard clam (primarily *R. philippinarum*) habitat in West Coast estuaries and these additions have been shown to enhance juvenile clam survival (Toba et al., 1992; Thompson, 1995). Thom et al. (1994) found that gravel addition to soft sediment significantly increased benthic respiration rates but had little effect on water quality parameters in south Puget Sound. They found site specific changes in surface macroalage, chlorophyll, and benthic assemblage, likely due to local conditions and time since the areas had been graveled. Secondary effects on the infaunal and epibenthic community were also shown to be site specific in later studies conducted in two sub-estuaries of south Puget Sound. Thompson (1995) found a general trend of enhanced abundance of gammaridean amphipods and nemerteans in modified substrate plots and reduced abundance of glycerid, sabellid and nereid polychaetes. Simenstad and Fresh (1995) documented site specific responses of the epibenthic

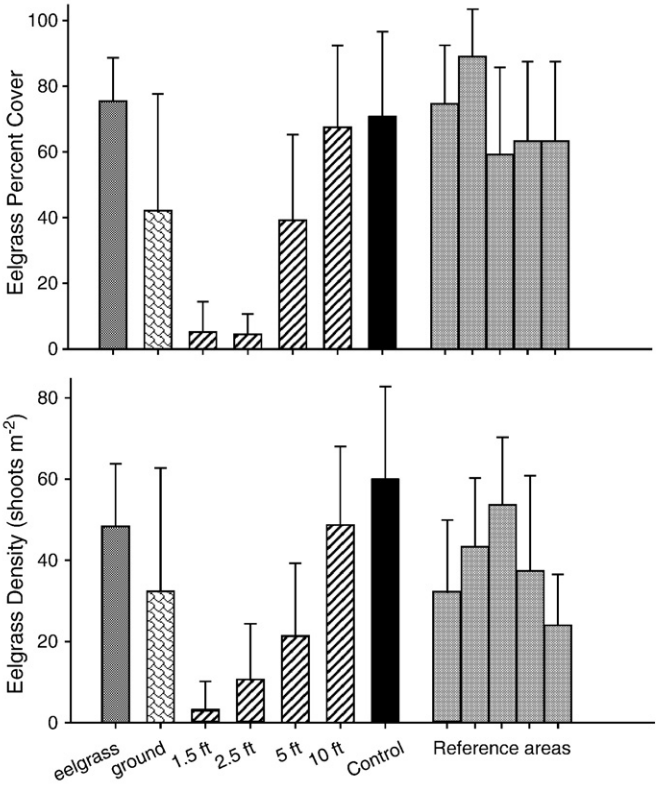

Fig. 6. Eelgrass spatial cover (top) and density of shoots (bottom) measured in experimental oyster long-line plots (1.5 to 10 ft spacing) in Humboldt Bay, California just before harvest at the end of the experiment compared to an eelgrass reference area nearby, ground culture plot (ground), a control plot with stakes but no lines (control) and 5 distant reference eelgrass beds (Mad River, Sand Island, East Bay, and Arcata Channel).

harpacticoid copepod community to the combination of gravel additions and predator exclusion netting.

Predator exclusion netting and/or bags are widely used for clam culture without substrate modification in West Coast estuaries and have been shown to increase the amount of organic carbon present in the sediment, likely due to biodeposition from larger age classes of *R. philippinarum* which were more abundant in netted plots at farmed sites in British Columbia, Canada (Munroe and McKinley, 2007a,b). Little consistent effect however was shown for sediment grain size or the abundance of other bivalves. Settlement of the cultured species *R. philippinarum* displayed highly significant interannual differences and the effect of netting was marginally significant (decreased settlement), but could be negative or positive depending on presence and size class of older clams (Munroe and Mckinley, 2007b; Whiteley and Bendell-Young, 2007). Finally, in Baynes Sound, British Columbia where some growers also installed beach fences around their plots, Zydelis et al. (2006) found densities and distribution of important wintering populations of surf and white-winged scoters to be primarily related to environmental factors and not shellfish aquaculture though 76 ha or 5% of the intertidal area was recorded to be covered by predator exclusion nets.

6. Harvest practices as pulse disturbances

Fisheries harvests in general can remove non-target species and re-set systems to early-succession conditions. However, the initial impact and pace of recovery clearly vary with harvest method, type of habitat present, and organism being studied (Kaiser et al., 2006). In their recent review, Kaiser et al. (2006) found just 6 examples of intertidal raking, which is perhaps most relevant to shellfish aquaculture (since intertidal dredging involved sediment removal and longer recovery times linked to infill rates; see also Dernie et al., 2003). Their meta-analysis showed that initial impacts to biota were relatively small and harvested areas matched controls within 50 days. Recovery was slower however in muddy sand and in biogenic habitats (especially when the latter included larger, older organisms such as corals and bivalves which contributed directly to biomass removed; see also Lenihan and Peterson, 2004). This is likely to be the case after disturbance to seagrasses which are sensitive to a variety of activities with some parallels to aquaculture harvest practices: dredge and fill (Fonseca et al., 1984; Onuf, 1994; Erftemeijer and Lewis, 2006), boat propellors (Zieman, 1976; Dawes et al., 1997), and boat anchor and mooring chain scars (Walker et al., 1989; Thom et al., 1998). Repeated trampling (mimicking recreational visitors) reduced the biomass of *Thalassia testudinum* in Puerto Rico, especially in softer substrates (Eckrich and Holmquist, 2000), but harvest activities for shellfish are unlikely to occur this frequently. Trawling and dredging for wild shellfish also negatively affect seagrass (Fonseca et al., 1984; Peterson et al., 1987; Orth et al., 2002; Neckles et al., 2005), although an extension to aquaculture must consider gear, technique, species ecology of seagrass (e.g. Erftemeijer and Lewis, 2006), and the physical environment (e.g. Kaiser et al. 2006). Several hard clam harvest methods have been shown to reduce eelgrass, including mechanical “clam kicking” with propeller wash (Peterson et al., 1987) and hand digging when rhizomes were extensively fragmented (Cabaco et al., 2005). The scale of harvest activity has also been shown to be important for both the direct effect on seagrass and associated organisms and the secondary impact of harvest on food for shorebirds and waterfowl. Small scale harvest of clams by hand in a national park in Spain (Navedo and Masero, 2008) appeared to have low impact and be sustainable, while larger scale effects of dredge harvesting on wild stocks of mussels and cockles in intertidal areas of the Dutch Wadden Sea are highly debated (Piersma et al., 2001; Verhulst et al., 2004; Kraan et al., 2007).

Recovery time after disturbance to seagrass should vary with seagrass species, disturbance size, disturbance intensity, and sediment characteristics. Seasonal time of disturbance is also likely a factor. Seagrass can recover via lateral rhizome spread or via sexual reproduction and seed dispersal depending on location and species. In fact, both natural and human disturbances have been shown to enhance sexual reproduction in seagrass (Marba and Duarte, 1995; Peterken and Conacher, 1997; Plus et al., 2003; Olesen et al., 2004). With respect to aquaculture, intertidal clam harvest in Portugal resulted in 2 fold higher seed production and an extended reproductive season for *Z. noltii* which enabled it to recover from harvest within a year (Alexandre et al., 2005).

Most of the research outlined above on press disturbances due to aquaculture in West Coast estuaries has not addressed the direct response of the benthic community to the pulse effect of harvest practices because it is not generally possible to distinguish these from effects of just adding the cultured organisms themselves. Consequently, the most valuable insight into harvest practices comes from before–after comparisons, which can then be tracked over time to determine pace of recovery. Both the initial impact and time to recovery have been variable in studies of the effect of oyster harvest to eelgrass on the U.S. West Coast. Results of experimental dredging using a toothed metal dredge at relatively large scale (0.33 ha plots) in Willapa Bay, Washington provide one explanation for this variation. At a muddy site, eelgrass initially declined 42%, where shoot and rhizome removal by the dredge implement was substantial, requiring 4 years for recovery, whereas at a sandy site, initial decline was only 15% and recovery occurred in 1 year (Tallis et al., in press). The effects of multiple passes with a suction dredge were evaluated by Wadell (1964) who found up to 96% initial loss of eelgrass biomass in Humboldt Bay with recovery taking up to 2 years. Treatment frequency also varies substantially and growers suggest that suction dredges are no longer common, each company designs their own mechanical implements, operator experience can be a factor, and mechanical harvest is rarely used in soft muddy sediments. In a comparison of eelgrass across three types of aquaculture (longlines, hand-picked, dredged), it proved possible to separate the effects of different culture practices from the effects of oysters, because oyster cover was included as a continuous variable ranging from <5% to >50% across beds. Relative shoot growth rates were 15% higher in both ground and long-line culture beds, but eelgrass production per unit area was driven by density and plant size differences and therefore lower in all aquaculture beds than in nearby eelgrass reference areas. Hand-picked beds had higher eelgrass production per unit area than did dredged beds (Tallis et al., in press), indicating higher impacts of mechanical harvest than picking up oysters by hand in eelgrass. For large areas such as aquaculture beds to regain eelgrass requires seed germination or asexual reproduction of remnant adults. In Willapa Bay, Washington seed germination can be high (>4 m^{-2}), particularly on dredged beds (Wisehart et al. 2007), although seedling survival appears universally low (1–2%; Wisehart 2006). Rhizome branching appears to be important for recovery of gaps in eelgrass (up to 16 m^2), but only occurs seasonally and thus gaps created experimentally in mid-summer did not begin to recover from the edges until the following spring (E.L. Wagner, unpubl. data). Clearly how much sexual versus asexual reproduction contributes to eelgrass resilience is important and may vary both temporally and spatially, but these dynamics have not been investigated on the U.S. West Coast.

For clams, effects of harvest appear related to the extent and depth to which sediment is dislodged. Effects of recreational clam harvest using rakes on *Z. marina* were undetectable, but digging clams with shovels reduced eelgrass cover and biomass over the short term, although recovery occurred fairly rapidly (months) in Yaquina Bay (Boese, 2002). Though the introduced seagrass, *Z. japonica* has expanded into areas and often now interferes with clam aquaculture on the West Coast of the U.S., clam aquaculture does not co-occur with *Z. marina*. Recreational clam harvest in the San Juan Islands, Washington caused short term impacts to non-target clam species abundance and polychaete species richness due to sediment displacement with shovels (Griffiths et al., 2006), but this does not typically occur for aquaculture where harvesting is typically done by hand or small rake and sediment replaced. In an experimental study of the effects of geoduck aquaculture on eelgrass density in south Puget Sound, Washington small (1 m^2) gaps in eelgrass beds required >1 year for recovery via regrowth from the edges, because flowering and seed germination were very rare (Ruesink and Rowell, in prep.). When the geoducks were harvested, eelgrass shoot density dropped >70% and recovery was subsequently difficult to gauge because control plots also declined in density over the 3-year study (Ruesink and Rowell, in prep.).

7. Landscape considerations

The available evidence discussed above for the U.S. West Coast indicates that some types of bivalve shellfish aquaculture can have effects on other species, and these effects may be place- and time-specific in part due to the scale at which observations are made. The vagueness of this conclusion is to be expected from ecological studies: unfortunately, it leads to the potential for selective use of evidence to support a conclusion of strong positive, strong negative, or weak effects of aquaculture. An important avenue of future research lies in documenting and understanding the role of aquaculture at an appropriate landscape scale, where aquaculture is intermixed (literally overlapping, as with eelgrass in oysters; or distributed as meadows and patches) with other habitat types.

There is no particular scale inherent in the concept of a landscape, only that it has a spatial dimension. For the purposes of this discussion however, we use a common definition of a spatially defined mosaic of heterogeneous elements that differ in their qualitative or quantitative properties (Wiens, 2005). We consider the estuarine landscape on which aquaculture acts as a disturbance and therefore define it to be larger than the scale of an individual lease, bed, reef, or set of structures used to culture shellfish. Conceptually this differs from estuary to estuary and is influenced by aquaculture practice and the cultural/political framework that exists in a given place. A series of questions that might then be asked regarding this landscape include (after Ahern, 2005):

> What is the proper spatial and temporal scale for understanding ecological patterns and processes in the estuarine landscape?
> How large a habitat patch (shellfish bed, eelgrass meadow) is required to support a given species or ecological process?
> Do these habitats form a “corridor” that connects larger habitat areas and if so what configuration of corridors is necessary to sustain species or ecological processes across the estuarine landscape?
> Which species or species group should be planned for? Can a particular “indicator” species represent the habitat needs of a group of species?
> Are there ecological interactions between shellfish aquaculture and other common anthropogenic disturbances at landscape scales?
> How does a particular estuary constrain or support an ecological process?
> Estuaries are open systems and connected and influenced by the nearshore coastal ocean and the watershed — how does this affect the ecological processes?
> How should aquaculture as a disturbance be understood in the estuarine landscape?
> Within cultural and economic constraints, can aquaculture be incorporated into estuary planning to lessen or enhance the potential effects to these other habitats and therefore species that utilize them?

These questions about the influence of habitat configuration on organism abundance and behavior at broad spatial scales (relative to the organism being studied) have been widely examined in terrestrial systems (Kareiva, 1987; Forman, 1995; Mazerolle and Villard, 1999; Debinski and Holt, 2000; Lindenmayer and Fischer, 2006), but only recently considered for marine habitats like eelgrass beds and oyster reefs (Brooks and Bell, 2001; Fonseca et al., 2002; Salita et al., 2003; Harwell, 2004; Darcy and Eggleston, 2005; Grabowski et al., 2005; Hovel and Fonseca, 2005; Bostrom et al., 2006; Connolly and Hindell, 2006; Johnson and Heck, 2006; Tanner, 2006; Hinchey et al., 2008). Increased connectivity between marine populations due to passive dispersal of larval stages and juveniles over large areas suggests that landscape scale processes differ in marine systems though there are clearly parallel processes to be explored. Corridors and habitat fragmentation have been shown to be less important, particularly for many invertebrates with pelagic larvae, but also for more sedentary adults (e.g. bivalves and small polychaetes, Bowden et al., 2001; Tanner, 2005; Cole et al., 2007). Fragmentation, patchy seagrass beds, and increased habitat edges may actually enhance diversity and increase the density of some bottom feeding invertebrates like decapod crustaceans and fish, whereas larger seagrass meadows may harbor higher numbers of smaller cryptic species (Salita et al., 2003; Tanner, 2005; Selgrath et al., 2007). Clearly other factors are also important like water depth, water movement, predation and organism behavior and motility (Irlandi et al., 1995; Darcy and Eggleston, 2005; Jackson et al., 2006; Horinouchi, 2007). Effects at the estuarine landscape scale are potentially more important for motile organisms with increased perception of structure at this scale and a greater home range which also provides important linkages between habitats like seagrass and

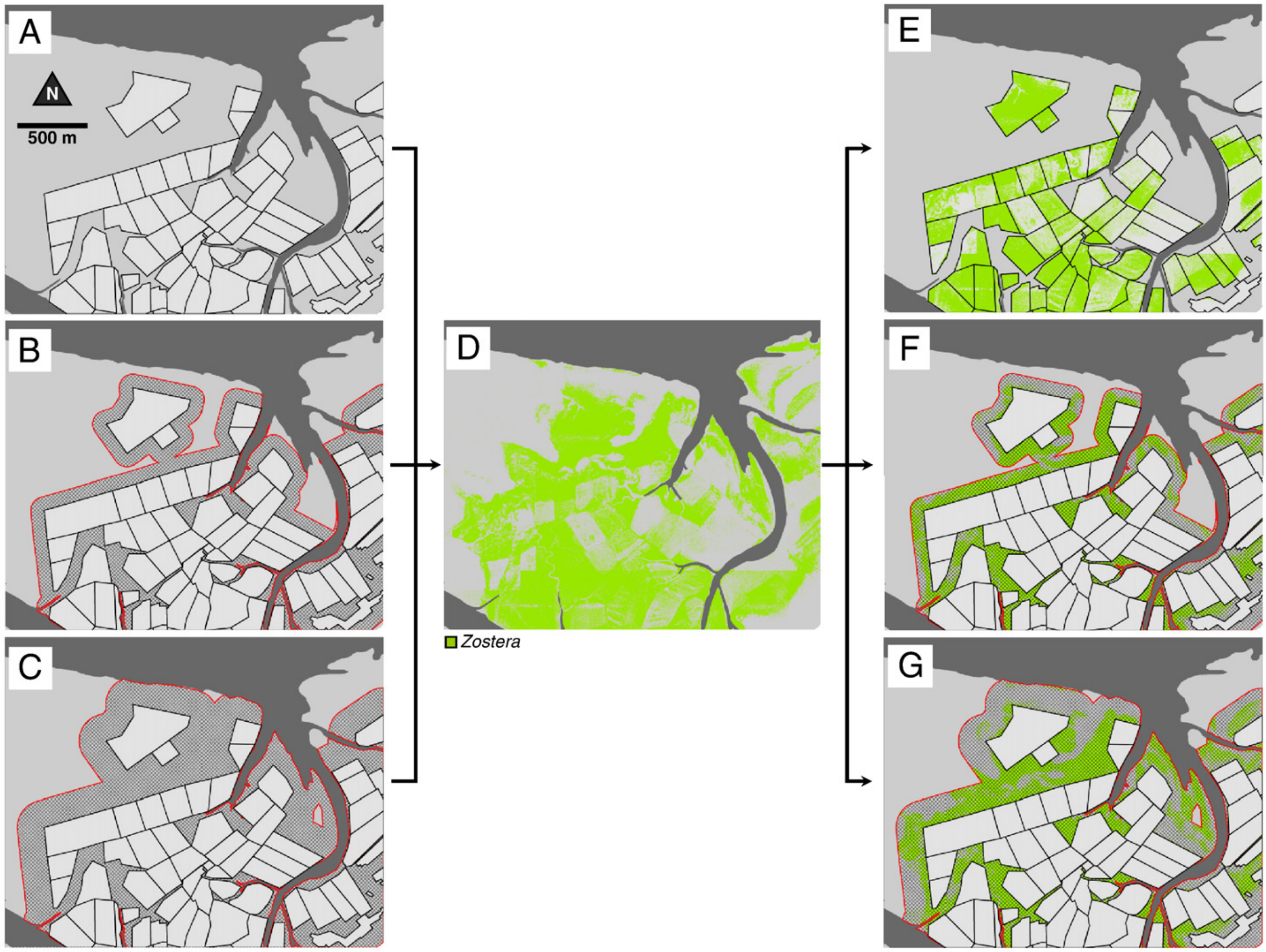

Fig. 7. Infrared aerial photography of an area near Stony Point in Willapa Bay, Washington was used to determine presence of intertidal vegetation (*Zostera* green shading, D). A separate layer was created which shows the distribution of active oyster aquaculture (A, top left) based on interviews with growers and both a 100 m and 200 m buffer zone around the edge of the culture areas (B and C). Estimates were then made of the proportion of the total area represented by *Zostera* in each of these zones (E, F, G). Although it represents a temporal snapshot (May 2005), the proportion of area covered by vegetation is comparable inside and outside aquaculture in these zones (46% inside aquaculture, 50% in 100 m buffer, and 44% in 200 m buffer).

marsh (Irlandi and Crawford, 1997; Simenstad and Cordell, 2000; Bostrom et al., 2006).

There have been few landscape-level approaches to bivalve shellfish aquaculture, although some progress has been made in Willapa Bay, Washington. Here, there are estimates of the total amount of different habitat types and how these have changed over time. In addition, the behavioral response of selected species (salmon, crab) has been studied across habitat types including bivalve shellfish aquaculture areas by these larger mobile organisms as discussed in Section 5.3 above (Pinnix et al., 2005; Holsman et al., 2006; Hosack et al., 2006; Semmens, 2008). Nonetheless, the influence of aquaculture has not yet been examined at a landscape scale on the West Coast of the U.S. and new work will need to be done to address such landscape-level features as patch size, connectivity, and the population response of organisms. Managers and regulators rightly suggested a general “no net loss” policy for estuarine wetlands which include eelgrass. This constraint has focused their efforts to date on protecting existing eelgrass as valued structured benthic habitat without much consideration of other forms of habitat or the location and scale of eelgrass habitat. Studies to date have also mostly examined organism presence and density in a given habitat and not broad scale spatial pattern or functional roles of these habitats. It could be that some habitats are more important than others at a broader landscape scale (e.g., as protective cover near channel edges for juvenile salmon) and that the configuration of both shellfish and submerged aquatic vegetation as habitat is also important because it provides food for larger more mobile organisms at that scale (e.g., for juvenile salmon, English sole, or shorebirds and waterfowl as discussed above) and protective cover and food for others (e.g for juvenile crab). Applications might be gleaned from work in the terrestrial environment where agricultural field margins and forest edges have been considered and managed as valuable habitat (New, 2005).

Despite the generally negative results of disturbance to eelgrass from aquaculture on small spatial and short temporal scales discussed above, eelgrass is generally present and intermingles with shellfish on all aquaculture beds at the tidal elevation where it is found naturally in Willapa Bay, Washington. Studies conducted to date have not evaluated historical records to indicate either loss or gains in eelgrass habitat over time, nor whether eelgrass would have been present regardless of subsequent aquaculture activity. Across Willapa Bay as a whole, shellfish aquaculture currently occupies about 13% of the estuary (4625 ha) and 20% of the tideflat (B.R. Dumbauld, unpubl. data; Feldman et al., 2000). It has likely historically replaced at least two other habitat types: monospecific eelgrass (*Z. marina*) and burrowing shrimp (*N. californiensis* and/or *Upogebia pugettensis*).

Interestingly, the application of carbaryl to remove burrowing shrimp may actually enhance eelgrass (both the native species and a non-native congener *Z. japonica*, Dumbauld and Wyllie-Echeverria, 2003). With the exception of changes in practices like switching from on-bottom culture to off-bottom culture in some locations, the press (oyster addition) and pulse (planting and harvest operations) disturbances of oyster culture have not changed materially for decades (Ruesink et al., 2006), so there is no reason eelgrass would necessarily be worse off now than in the recent past. Indeed, there is scientific evidence that eelgrass fluctuates with environmental conditions (Thom et al., 2003) and compelling evidence that it has been expanding its distribution in Willapa Bay (Ruesink et al., in review) and other estuaries along the open coast of the western U.S. even though it is declining elsewhere in the world (Orth et al., 2006) and in isolated locations on the U.S. West Coast (e.g. Hood Canal and San Juan Archipelago in Puget Sound, Gaeckle et al., 2007; Mumford, 2007). Based simply on tidal elevation, Willapa Bay was estimated to contain 3139 ha suitable for *Z. marina* (0 to −1.2 m MLLW) in the 1850 s, increasing to 4845 ha in the 1950s as the bathymetry became shallower (Borde et al. 2003). A recent estimate based on aerial photography (3424 ha) may be slightly lower than the potential area because other habitat types (e.g. aquaculture, burrowing shrimp) also occur at the same elevations (Ruesink et al. 2006). The two *Zostera* species together may cover 4935 ha (B.R. Dumbauld, unpubl. data) or 6162 ha (Ruesink et al. 2006). Despite their chemical control for aquaculture, burrowing shrimp currently also occupy a very large intertidal area in Willapa Bay (3060 ha = 13.5% of the intertidal; B.R. Dumbauld, unpubl. data). They have probably fluctuated in abundance and would have the potential to occupy a much larger area if shrimp control had not occurred, with attendant effects on both native and non-native species of *Zostera* and associated benthic community. Simenstad and Fresh (1995) estimated that 12.6% of an area near Stony Point in Willapa Bay was highly disturbed due to aquaculture with little to no eelgrass present. Despite the obvious signature from oyster culture disturbance however, when we re-examined the proportion of area with eelgrass present in oyster culture beds and compared it to that area at a similar tidal elevation just outside oyster culture (where eelgrass would be expected to occur) in the same Stony Point vicinity (2005 data, Fig. 7), we found little difference in eelgrass cover (46 versus 50%). Such comparisons are merely one-time snapshots of the presence of vegetative cover and more thorough analyses of a larger area over a slightly longer temporal scale with more descriptive categorizations of bed use will reflect the importance of vegetative recovery processes and perhaps other details discussed in above sections. In contrast, historical estimates do not account for occupation by other species, for instance, based on crude maps from the late 1800's, native oyster *O. lurida* beds occupied 2700 ha (12% of the low intertidal and shallow subtidal) that now consists of relatively undisturbed and dense native eelgrass meadows (Collins, 1892; Townsend, 1896; Sayce, 1976). Thus shell habitat has always been present in Willapa Bay, although its current distribution is at a higher tidal elevation than would be assumed from a contemporary projection, or than is present at other important West Coast locations (Tables 2 and 3). These spatial analyses have rarely been conducted in West Coast estuaries (but see Carswell et al., 2006 for Baynes Sound, British Columbia, Canada and Ward et al., 2003 for Bahia San Quentin, Mexico).

8. Resilience — management implications and research needs

Shellfish aquaculture has been an important activity and has supported local economies along the West Coast of the U.S. for at least 100 years. At present temporal and spatial scales in West Coast estuaries, our review suggests that the practice of shellfish aquaculture viewed as an ecological disturbance seems much more sustainable than other human activities such as freshwater diversion, coastal development and pollution, which continue to degrade estuarine function. On a global scale however, aquaculture is expanding and so may pressure to increase regional use of estuaries for bivalve culture. We end this review with some conclusions, caveats, and research needs which we hope will be useful to managers and decision-makers.

Table 3
Status of aquaculture in some U.S. West Coast estuaries and a possible classification system (after Weinstein and Reed, 2005) which could be used as a starting point for discussing sustainability and future planning.

Estuary	Area (km^2)	Existing aquaculture (h)	Proportion	Types	Other anthropogenic disturbances in order of importance	Possible classification zones
Baynes Sound, British Columbia, Canada	87	458	0.053	Oyster — bottom–deepwater clams	Nutrients	Zones: Production, Conservation
Totten Inlet, Puget Sound WA	24.7	85	0.034	Clams, geoducks, oysters on bottom, racks, off-bottom	Nutrients,	Production/conservation
Grays Harbor, WA	255	364	0.014	Oyster — longline, bottom	Nutrients, dredging	Zones: Production, urban-industrial
Willapa Bay, WA	358	4626	0.129	Oyster— longline, bottom clams	Marsh fill	Production
Tillamook Bay, OR	37.3	1014.8	0.272	Oyster — bottom, long-line	Nutrients, marsh fill	Production
Netarts Bay, OR	11.1	154.2	0.139	Oyster — bottom	Nutrients	Production/conservation
Salmon River, OR	1.8	0	0	Na	Marsh fill	Conservation
Siletz Bay, OR	5.9	0	0	Na	Nutrients	Conservation
Yaquina Bay, OR	17.6	210	0.119	Oyster — raft, on bottom	Marsh fill, nutrients, dredging	Zones: Urban industrial, production/conservation
Alsea Bay, OR	10.2	0	0	Na	Nutrients	Conservation
Coos Bay, OR	53.8	97.3	0.018	Oyster — bottom, long-line	Nutrients. Marsh Fill, Dredging	Zones: Production, conservation, urban-industrial
Humboldt Bay, CA	67	121.4	0.018	Oyster — longline	Marsh Fill, Nutrients	Zones: NB = production SB = conservation
Drakes Estero, CA	9.2	12.1	0.013	Oyster — racks and bag on bottom	Nutrients	Conservation/production
Tomales Bay, CA	28.5	240	0.08	Clams	Nutrients	Production
San Francisco Bay, CA	1060	0	0	Historical oyster harvest	Diking and fill, Modified FW flow, nutrients	Zones: Urban industrial, conservation

Aquaculture numbers represent estimates of actual ground used for culture as determined from grower interviews for Willapa Bay and Grays Harbor, WA and actual fingerprint used for California estuaries (Tom Moore, Calif. Dept. Fish and Game).
In contrast, figures include total area leased from the state for Oregon estuaries (John Byer, Oregon Dept. of Agriculture) and total owned and/or leased ground for Totten Inlet (Dan Cheney, Pacific Shellfish Institute), and total tenures from Carswell et al. (2006) for Baynes Sound. Thus proportion of estuary occupied by culture for some estuaries is likely over-estimated. Estuarine areas for Oregon are from (Cortright et al., 1987).

From a manager or land-use planner's perspective, the first consideration in evaluating shellfish aquaculture in a given estuary should be an answer to the question: What are we and/or should we be managing for? Estuaries have a wide range of potential functions, have been and will continue to be influenced by many human activities, and similarly are influenced by many natural disturbances in addition to shellfish aquaculture. While the current paradigm for most managers is whole “ecosystem based” management (Grumbine, 1997), in reality managers have only progressed to varying degrees down this path, especially for marine systems. Thus the answer to “what are we managing for?” is driven by a wide variety of stakeholders and societal values (social historical, political, moral and aesthetic as well as economic; Leslie and McLeod, 2007; Weinstein, 2007; Ruckelshaus et al., 2008). Although these values are outside the purview of our intended review, we found it instructive to at least classify West Coast estuaries by the current level of aquaculture and other anthropogenic disturbance as Weinstein (2007) propose. Willapa Bay and Humboldt Bay might therefore be considered “production” estuaries with greater than 10% of the area occupied by shellfish aquaculture, while numerous other smaller estuaries with little aquaculture could be classified as other types (though the proportion of total estuarine area leased for aquaculture in some of these systems is also greater than 10%, leased does not necessarily mean actively used, Table 3). Portions of estuaries might also be classified or zoned separately this way (e.g. the South Slough portion of Coos Bay, Oregon or South Bay portion of Humboldt Bay, California). These classifications would then further help set goals and priorities for management and restoration, an activity which also involves social decisions about what should be “restored” (Simenstad et al., 2006). Classifications of production and production/conservation would be compatible with sustainable aquaculture, whereas areas classed as conservation or conservation marine within estuaries might not include aquaculture depending on the level of anthropogenic influence and goals for sustaining traditional commercial and recreational products desired. These decisions would obviously vary by jurisdiction. In Washington state for example, a critical societal decision was made in 1895 with passage of the Bush and Callow Acts which deeded 18,932 h of tidelands to private ownership specifically for the purpose of commercial shellfish culture. This set the stage for continued industry involvement and emphasis as a priority activity. Within such a framework which simply recognizes the current status and constraints on these systems, we offer the following conclusions specific to bivalve aquaculture as disturbance, its relevance to resilience in West Coast estuaries, and suggestions for future research:

1. Bivalves process phytoplankton and alter the forms and distribution of nitrogen in a system. In typical U.S. West Coast systems evaluated to date, water column and sediment nutrient concentrations are generally relatively high and greatly influenced by the proximity to deeper nearshore ocean waters where upwelling controls production during summer months. The situation may be different for small systems such as coastal lagoons or portions of large fjords like Hood Canal in Puget Sound, Washington where circulation is restricted. Very little modeling of whole-system energy and nutrient budgets, including aquaculture, has been done regionally, although the methods are well worked out in Europe. We suspect, however, that terrestrial and anthropogenic nutrients will figure less prominently than in many other places where bivalves are grown. Studies that expand on work like that completed for Willapa Bay, Washington showing the potential for bivalves in one part of the estuary to limit production in another part (Banas et al., 2007), and comparisons with other systems including portions of fjords like Puget Sound, Washington would be extremely useful research avenues. Intermediate bed scale studies such as those conducted using flow models and benthic nutrient flux estimations within given estuaries (Newell et al., 1998; Porter et al., 2004) will still be necessary to calibrate the larger landscape scale estimations, particularly with new species or culture techniques (e.g. geoducks in tubes, oysters on longlines).
2. Some bivalves and culture practices modify estuarine habitat at local community and at landscape scales. The effect of aquaculture is most often evaluated against existing structured habitat in the form of submerged aquatic vegetation. While bivalve aquaculture might be viewed as a press disturbance over the long term in a given area, the individual activities act as pulse disturbances and *Z. marina* in U.S. West Coast estuaries can recover to pre-disturbance levels relatively rapidly (within a period of 2 years in some systems). This is usually before the next planting or harvest disturbance occurs, but depends on conditions and the aquaculture practice (e.g. oyster fattening beds might be rotated on a yearly basis and thus disturbance is frequent, while seed-harvest beds are left undisturbed for 2 to 4 years). Furthermore, the extent of the effect depends on the practice (hand harvest versus dredge harvest, longlines versus on-bottom culture). The current distribution of eelgrass reflects a balance of space competition, pulse disturbance and recovery, and is therefore at dynamic equilibrium on aquaculture beds (albeit generally lower than in undisturbed eelgrass meadows). Research is still needed on factors that cause plants to alter their reproductive strategy and enhance seedling production (Wisehart et al. 2007), whether plants respond differently to disturbance across seasons, particularly since densities vary naturally over the year (Ruesink et al., in review) and finally on the effect this has at larger spatial scales (growing areas to estuary) and over relevant temporal scales (at least the lifetime of a shellfish crop = 3 or 4 years).
3. The role of aquaculture (organisms themselves and support structures) as estuarine habitat should also be considered. For small benthic infauna and mobile epibenthic fauna, structure provided by aquaculture appears functionally similar to eelgrass, based on invertebrate abundances and composition measured to date in West Coast estuaries. For larger benthic invertebrates and fish, use of habitat depends on mobility and varies with life-history stage and taxon being evaluated, so temporal and spatial scales are important considerations. Though less is known about habitat function for these larger more mobile organisms, they can use structure for protection from even larger predators (juvenile salmon in eelgrass and 0+ Dungeness crab in oyster), but still rely on other habitats for foraging (1+ Dungeness crab in unstructured open habitat). Given the presence of mixed habitats (i.e. eelgrass within aquaculture beds), it would be valuable to determine relationships between eelgrass density and its ecosystem function, effective habitat patch sizes, and corridor use at a larger landscape scale. This may be an area where best management plans can be designed and implemented since the shellfish industry would likely be supportive of maintaining habitat corridors (e.g. eelgrass along channel edges) and timing windows (e.g. limited harvest operations in some areas during fish spawning or bird migration periods) should they prove necessary.
4. Finally, it is important to consider estuarine changes not simply in terms of departure from baseline, but as they influence resilience, that is, capacity of the system to withstand or recover from other shocks. Aquaculture as disturbance is generally within the scope of existing “natural” disturbances to the system (e.g winter storms) and other ecosystem engineers (e.g. eelgrass and burrowing shrimp) are also inherently adapted to this scale of disturbance. Certain anthropogenic disturbances have reduced estuarine resilience, for instance habitat removal via wetland diking and filling, hardening of surfaces in the watershed, nutrient additions, invasive species such as *Spartina*, and possibly food web modifications like removal (sharks, skates and sturgeon) or protection (harbor seals and sea lions) of large predators. In contrast, bivalve aquaculture does not remove area from the estuary or degrade water quality,

and thus is less likely to undermine resilience. Though local and short term effects are clearly evident in U.S. West Coast estuaries, bivalve aquaculture has not been implicated in shifts to alternate states or reduced adaptive capacity of the larger ecological system. Typical thresholds that might be involved in such catastrophic change would likely be reached first with other human disturbances (e.g. nutrients and predator removal), although location and scale remain important management considerations (e.g. small inlets with stratified water columns and less routine physical disturbance might exhibit lower thresholds to large scale aquaculture operations).

Acknowledgements

The authors sincerely thank numerous contributors to aquaculture studies in Willapa Bay, Puget Sound and Humboldt Bay including especially Alan Trimble, Lee McCoy, Sally Hacker, Lorena Wisehart, Victoria Poulton, Geoff Hosack, Heather Tallis, Bruce Kauffman, Beth Wheat, Kirsten Rowell, Kirsten Holsman, Dan Cheney, and Andy Suhrbier. Lee McCoy was instrumental in conducting mapping efforts and producing GIS results for Willapa Bay. Shellfish growers in all of these locations provided access to their tidelands to conduct studies, logistical support, and invaluable practical insight about local conditions and history. Funding for these studies was provided by the Western Regional Aquaculture Center (2001-38500-10495 and 2003-38500-13198), Grant #NA 16R1044, Project R/ ES42 from NOAA to Washington Sea Grant, the Andrew W. Mellon Foundation, Washington Dept. of Fish and Wildlife, and USDA-ARS. The authors also thank Gary Banowetz, Dan Cheney, Chris Langdon and three anonymous reviewers for their comments on the manuscript.

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Aquaculture 498 (2019) 254–262

Contents lists available at ScienceDirect

Aquaculture

journal homepage: www.elsevier.com/locate/aquaculture

Bivalve aquaculture and eelgrass: A global meta-analysis

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ARTICLE INFO

Keywords:
Aquaculture
Bivalve
Shellfish
Eelgrass
Zostera
Meta-analysis

ABSTRACT

The marine, intertidal zone is the optimal environment for eelgrass (*Zostera* spp.) and bivalve aquaculture. Eelgrass is a valuable and protected nearshore habitat. It is important to understand how bivalve aquaculture interacts with eelgrass to support the sustainable development of this globally expanding industry. This study provides a comprehensive understanding of the positive and negative effects of bivalve aquaculture on eelgrass by conducting the first quantitative, global meta-analysis of aquaculture-eelgrass studies. A literature review resulted in 125 studies that met established criteria for inclusion in this analysis. The meta-analysis determined: (1) how eelgrass responds to on-bottom and off-bottom bivalve aquaculture, (2) how these responses vary between regions and specific grow-out methods, and (3) the resilience of eelgrass after harvesting disturbances. On-bottom culture (laying directly on the sediment potentially including predator exclusion devices) corresponded to significant increases in eelgrass growth and reproduction, and a decrease in density and biomass. Off-bottom culture (e.g., longline and suspended bag) resulted in significant decreases in eelgrass density, percent cover, and reproduction. Results support a space-competition hypothesis for on-bottom culture and provide limited support for light limitation in off-bottom culture, although other mechanisms of interaction are potentially occurring as well. A US west coast case study revealed regional differences in eelgrass responses, including a more negative trend in eelgrass density from off-bottom culture, and a neutral effect on reproduction from on-bottom culture (relative to neutral and positive trend, respectively, in the average of all other studies). Eelgrass densities recovered after all harvest methods, however mechanical harvest methods created greater initial impact and longer recovery times than manual harvest methods. The time-period over which observations were reported was an important variable that was not included in the analysis but could influence these results. These analyses suggest the response of eelgrass to bivalve aquaculture varies depending on eelgrass characteristics, grow-out approaches, and harvesting methods, with potential regionally specific relationships. Questions remain, regarding how this dynamic relationship between eelgrass and aquaculture habitat relates to ecological functions and services in the nearshore environment.

1. Introduction

Eelgrass (*Zostera* spp.) has physiological and substrate requirements similar to those of cultivated bivalves (class Bivalvia), resulting in overlapping distributions and causing concerns over the expansion of aquaculture in coastal estuaries (Cullen-Unsworth and Unsworth, 2013; Seitz et al., 2014). Eelgrass is a valuable nearshore habitat that provides numerous ecosystem services and functions, such as primary productivity, nursery habitat, sediment stabilization, predator refuge, and carbon sequestration (Jackson et al., 2001; Duarte, 2002). Despite its ecological importance, widespread threats to eelgrass habitats are well documented (Lotze et al., 2006; Orth et al., 2006) and analyses suggest declines in many eelgrass populations (Waycott et al., 2009, but see Shelton et al., 2017). Interactions between eelgrass and bivalve aquaculture have been proposed as a potential driver of eelgrass declines (Rumrill and Poulton, 2004, Tallis et al., 2009, but see Forrest et al., 2009, Dumbauld and McCoy, 2015). However, interactions between bivalve aquaculture and nearshore marine communities remain poorly understood and are challenging to summarize across studies due to variation in experimental design, species being cultured, grow-out methods, harvest and maintenance disturbance regimes, scale of development, and local environmental conditions. Qualitative reviews

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https://doi.org/10.1016/j.aquaculture.2018.08.046
Received 3 June 2018; Received in revised form 20 August 2018; Accepted 21 August 2018

have summarized existing studies (Dumbauld et al., 2009; Forrest et al., 2009), but they have not determined which trends are supported by the weight of the data, accounting for differences in experimental design, variation, and sample size.

Molluscan shellfish aquaculture (of which bivalves account for the vast majority) is an expanding, global industry that produced 16 million tonnes (US$19 billion) from marine and coastal habitats of America, Africa, Asia, Europe, and Oceania in 2014 (FAO, 2016). Bivalves (clams, oysters, mussels, and scallops) are grown in the nearshore marine environment, using a variety of grow-out and harvesting methods. Grow-out methods range from on-bottom (growing shellfish directly on the substrate, sometimes including predator exclusion devices) to off-bottom methods suspending bivalves above the sediment, including long lines, racks, stakes, and bags. Bivalves are harvested using manual (hand, rake, and hoe) and mechanical methods (digging, dragging, dredging, and sediment liquefaction). Cultivation and harvesting methods are often selected based on regional practices, substrate type, scale of operations, and environmental conditions. As regional management entities develop and evolve their aquaculture policies, there is an increasing request for a mechanistic understanding of how individual bivalve aquaculture practices influence eelgrass presence as well as eelgrass service and functions (Costa-Pierce and Bridger, 2002; National Research Council, 2010a; Coen et al., 2011).

The commercial cultivation of bivalves has coexisted with eelgrass for 100's of years, and bivalves have been harvested from these environments for 1000s of years prior (Mackenzie et al., 1997). Interactions between bivalve aquaculture and eelgrass exist, yet there is little established consensus on the overall trends and underlying mechanisms explaining how aquaculture and eelgrass interact (as reviewed in Dumbauld et al., 2009, Forrest et al., 2009, National Research Council, 2010a). Off-bottom culture can lead to light limitation (shading), potentially limiting eelgrass density, growth, productivity, and canopy structure (Burdick and Short, 1999, Rumrill and Poulton, 2004, Wisehart et al., 2007, but see Forrest et al., 2009). Conversely, increased sediment stabilization and wave attenuation from the addition of aquaculture-related structure could facilitate increased eelgrass percent cover and density (as reviewed in Dumbauld et al., 2009, McKindsey et al., 2011). On-bottom grow-out methods are hypothesized to compete for space with eelgrass, with potential to decrease eelgrass density (Tallis et al., 2009; Wagner et al., 2012). Increased eelgrass reproductive effort is an established response to disturbance, although the response of asexual versus sexual reproductive effort is less well understood (Alexandre et al., 2005; Cabaco and Santos, 2012; Ruesink et al., 2012). In eutrophic conditions, plankton-grazing bivalves can increase water clarity and, indirectly, enhance eelgrass growth (Newell, 2004). A combination of varying environmental conditions and cultivation methods can influence direct and indirect interactions between bivalve aquaculture and eelgrass, making the mechanism of interaction difficult to identify (Olesen and Sand-Jensen, 1994; Booth and Heck, 2009; Yang et al., 2013).

Multiple literature reviews have summarized existing studies on bivalve aquaculture and eelgrass interactions, but have not been able to provide insight on the weight of often-conflicting evidence (Dumbauld et al., 2009; National Research Council, 2010b; Coen et al., 2011; Mach et al., 2015; Herbert et al., 2016). A quantitative meta-analysis of shellfish aquaculture-ecological interactions can provide quantitative generalizations based on combining the data from independent studies (Englund et al., 1999). Meta-analyses have been broadly used in ecology and can be applied to aquaculture studies that follow an experimental design, including controls, treatments, and replicates (Hedges et al., 1999). Meta-analyses provide a formal method for combining information across studies and as a result can overcome limitations of variations in response metrics, scales of response, within study sample sizes, and levels of uncertainty. These analyses calculate a mean effect size for a given aquaculture-eelgrass interaction across multiple studies, accounting for inter-study heterogeneity, to identify shared responses among studies. Meta-analysis has the advantage over qualitative summaries that sum the number of studies supporting or refuting a hypothesis, in that it has statistical power, is less biased, and accounts for differences in study precision (Hedges and Pigott, 2001). Disadvantages to this approach include publication and research bias (e.g., significant findings are more likely to be published), incomplete data reporting (i.e., not reporting sample size and variance), lack of independence in effect size estimates, and bias in selecting studies to include in the analysis (Englund et al., 1999; Gurevitch and Hedges, 1999).

To provide essential knowledge for resource managers and the shellfish industry, we perform the first global meta-analysis of eelgrass-shellfish aquaculture interactions. We focus on three specific components: (1) how eelgrass responds to on-bottom and off-bottom shellfish aquaculture; (2) how these responses vary between grow-out gear types and region; and (3) the resilience of eelgrass after harvesting disturbances. Within these analyses, we test the hypotheses of light limitation caused by off-bottom shellfish aquaculture and space competition caused by on-bottom shellfish aquaculture. Finally, we use a US west coast case study to determine if regional patterns in aquaculture-eelgrass relationships are consistent with global averages.

2. Methods meta-analysis

2.1. Literature review & study selection

We conducted a literature review in the Web of Science using the following terms, where an asterisk symbol indicates any group of characters can be added to the end of that word:

1. eelgrass OR seagrass* OR zostera
2. AND aquaculture OR culture* OR harvest* OR cultivat* OR farm*
3. AND shellfish OR clam* OR Ruditapes OR oyster* OR crassostrea OR Ostreidae OR mussel* OR mytilus OR geoduck* OR Panopea OR scallop OR bivalv*

Of the resulting list of publications, we selected studies that examined the relationship between eelgrass and shellfish, reported original data, had a study design that included a treatment and control, and reported uncertainty (standard error or standard deviation). Most studies included in our analysis related to cultured shellfish. We added a few that experimentally manipulated shellfish-eelgrass interactions outside of an aquaculture setting to increase our sample size (e.g., Reusch et al., 1994; Reusch and Williams, 1998). We used “Web Plot Digitizer” (http://arohatgi.info/WebPlotDigitizer/app/ accessed Oct. 25, 2017) to extract data from figures when they were not provided in the paper. When selecting publications to include in this study, we were as inclusive as possible to avoid introducing bias to the analysis (Englund et al., 1999).

We first divided studies into two disturbance types for separate analysis: those that examined how eelgrass responds to shellfish presence/absence, and studies that examined the response of eelgrass to shellfish harvest. We then separated these studies into six categories of eelgrass response metrics to maximize sample size: density, biomass, growth, percent cover, reproduction, and structure (Table 1, Supplementary Table A.1). The structure metric includes physical characteristics of eelgrass, such as blade width, leaf area, and height (Table 1). If a study reported data from multiple eelgrass response categories (e.g., eelgrass biomass and density), we included all the data, as each eelgrass response category was analyzed in a separate model. If a study reported multiple eelgrass metrics that fell under the same eelgrass category (e.g., eelgrass structure), we selected the one metric that was the most inclusive from that study (e.g., leaf area instead of leaf width or height). If a study included multiple treatment locations, we included all sites (accounting for the lack of independence by including “study” as a random effect in the analysis). If a study had multiple treatment levels

Table 1
The six eelgrass response metrics used in the meta-analysis and their corresponding eelgrass characteristics as they were reported in the literature.

Eelgrass response metric	Eelgrass characteristic
Density	Density
	Abundance
	Terminal shoot density
Biomass	Biomass
	Above-ground biomass
	Biomass ratio
	Shoot biomass
Structural	Blade width
	Canopy height
	Height
	Leaf area
	Leaf area index
	Leaf length
	Leaf width
	Number leaves
	Sheath length
	Shoot height
	Shoot size
	Leaf size
Percent cover	Percent cover
Growth	Growth
	Growth rate
	Leaf growth rate
	Shoot growth
	New leaf tissue
Reproduction	Density reproductive shoots
	Flowering shoots
	Reproductive effort
	Reproductive shoots
	Seed production
	Seed density
	Branching frequency
	Rhizome growth
	Rhizome biomass
	Rhizome diameter
	Rhizome internode length

(e.g., multiple levels of longline spacing, nutrient concentrations, or light availability) we selected the conditions that matched realistic local conditions where possible (often identified within the study). In studies that included multiple shellfish densities, we selected treatments of higher but realistic densities to better identify possible trends in eelgrass responses.

We restricted the eelgrass biomass category to above-ground biomass as we had limited data on below-ground biomass and reason to expect that metric would react differently to shellfish aquaculture (Yang et al., 2013). The eelgrass reproduction response metric included data on sexual and asexual reproduction, although we found limited data on the latter. While there is evidence that these metrics can respond differently to shellfish aquaculture (e.g., Ruesink et al., 2012), we kept both metrics to maximize sample size and found no change in overall significance of results when removing asexual reproduction. In the shellfish presence/absence analysis, we only used the final time point of reported time series to avoid issues of data dependence and focus on the longer-term trends in eelgrass. We included full time series data on the separate harvest–recovery analysis and as part of the length of study analysis described below. We only used studies that reported a non-zero variance in order to use the inverse variance weighting method described below.

2.2. Data preparation

The eelgrass response data were standardized using natural log-transformed response ratios (lnRR) to calculate effect size and variance (Eq. (1): Hedges et al., 1999)

$$\ln\mathrm{RR} = \ln\left(\frac{\overline{X}_T}{\overline{X}_C}\right) \tag{1}$$

here, $\overline{X}_T$ is the mean treatment response metric for eelgrass that co-occurs with bivalve aquaculture and $\overline{X}_C$ is the mean control response metric for eelgrass in undisturbed eelgrass beds, within a given study. The lnRR measures proportionate change between treatment and control groups such that positive values indicate higher values of the response variable (e.g., eelgrass density) in the experimental (bivalve aquaculture) treatments relative to the control treatments (eelgrass beds). Negative lnRR values indicate the experimental treatments had lower values than the control treatments. Values close to zero indicate little or no effect. If a treatment mean and treatment standard deviation equaled zero, we created new non-zero treatment values by multiplying the control mean and control SD by 0.1 to produce a low but valid number for lnRR calculations.

Variance (v) of the lnRR from a given study (k) is calculated using the standard deviation (sd), sample size (n), and mean response from the treatment ($\overline{X}_T$) and control ($\overline{X}_C$) (Eq. (2): Hedges et al., 1999).

$$v_k = \frac{sd_T^2}{n_T * \overline{X}_T^2} + \frac{sd_C^2}{n_C * \overline{X}_C^2} \tag{2}$$

We used mixed effect models in the meta-analysis to estimate the mean effect size in each analysis (Hedges et al., 1999; Koricheva et al., 2013). The random effect accounts for between study variation (residual heterogeneity). Studies were weighted by their variance to account for unequal sample sizes and variance. Studies with higher levels of replication, and thus more precise, were more heavily weighted and contributed relatively more to the overall mean lnRR of an eelgrass response metric. Following the random effects meta-analysis approach, the lnRR from each study (k) was multiplied by a weight (w) of the inverse of the sum of within study variance (v_k) and between study variance ($\hat{\sigma}^2$), as obtained from the mixed effect model (Eq. 3: Lajeunesse, 2011, Koricheva et al., 2013).

$$w_k = \frac{1}{v_k + \hat{\sigma}^2} \tag{3}$$

The weighted means of the individual studies were then used to estimate the grand mean effect size across all studies in the models described below. Final mean lnRR estimates were back-transformed to provide mean percent change for each gear/region/eelgrass response category $(-(100 - (\exp(lnRR) * 100)))$.

2.3. Shellfish presence/absence analysis

We used a meta-analytic approach to estimate the mean and variance of the distribution of effect sizes obtained from these shellfish-eelgrass studies (Hedges et al., 1999). The mean effect size of interactions between shellfish presence and the six eelgrass response metrics were calculated using mixed effect models in the Metafor package in the R programming environment (Viechtbauer, 2010, R Core Team, 2016). Each eelgrass response metric (biomass, growth, percent cover, density, reproduction, and structure) was modeled separately. In these models, “study” was included as the random effect to account for data obtained from multiple locations within some studies (20–40% of the studies used in each metric-specific model). The percent cover model did not include any studies with multiple treatment locations and was modeled without the random effect. First, we tested the overall effect of shellfish aquaculture on eelgrass by grouping all gear types and regions together. We then examined whether eelgrass response metrics varied between on-bottom culture (oysters and clams on/in the sediment with no associated predator exclusion devices) and off-bottom culture (oysters in longlines, stakes, racks, and suspended bags), including the on- and off-bottom grow-out category as a fixed effect. To determine if regional differences influence shellfish-eelgrass interactions, we included two regional categories based on sample size: US west coast (i.e.,

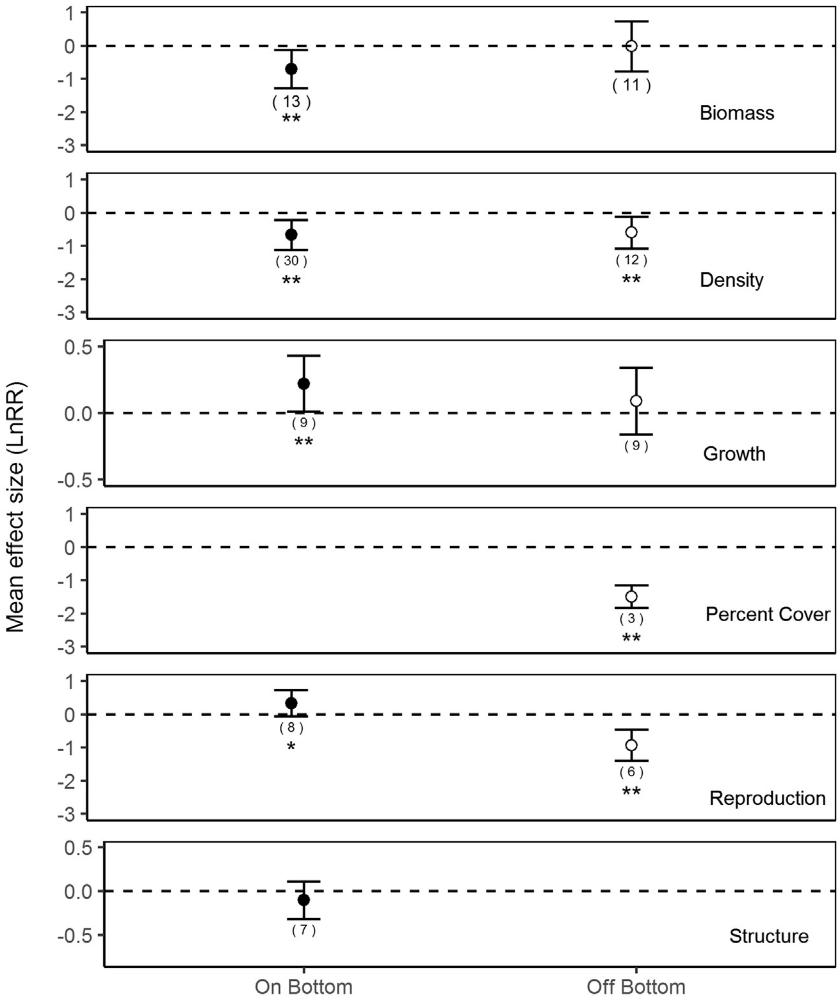

Fig. 1. The mean effect (95% confidence intervals) of on-bottom and off-bottom bivalve aquaculture on separate analyses of eelgrass biomass, density, growth, percent cover, reproduction, and structure. Sample size is in parenthesis and significance is indicated by * (< 0.1) or ** (< 0.05). The zero dashed line indicates no effect.

the Pacific coasts of Washington, Oregon, California states, Puget Sound, WA, and three studies from the west coast of Canada, and Mexico) and Global (the remaining studies excluding the US west coast). A third analysis examined eelgrass responses to various on- and off-bottom gear types. Results were only reported if the subgroup had a sample size of three or more studies. We considered results significant at a *p* value less than 0.1.

2.4. Harvest disturbance analysis

Studies identified as having a harvest disturbance were included in a separate analysis. We used mixed effect models to analysis change in eelgrass density over time (post-harvest). We used the same weighted, lnRR effect size and variance calculations described above, but this time included all time points in the published time series. In this analysis the random effect was "study" accounting for the measurement of eelgrass density at multiple time points within the same study. We restricted this analysis to the eelgrass density response metric, as sample size was limited for the other eelgrass response metrics across numerous harvest methods. Harvest methods were divided into Manual (hand, hand blade, hoe, and rake) and Mechanical (digging, dragging, dredging, sediment liquefaction) groups for analysis. We determined recovery as the time when there was no difference (lnRR = 0) between treatment (eelgrass in a site harvested for shellfish) and control (undisturbed eelgrass). We log-transformed time (days) to account for the large range in time scales across studies and harvest methods. We used an ANOVA test to determine significant differences between the manual and mechanical harvest groups slope and intercept.

2.5. Sensitivity analysis

Two potential biases in meta-analyses includes publication bias, in which non-significant studies are less likely to be published, and the disproportionate contribution of an individual study (Gurevitch and Hedges, 1999). We calculated Rosenberg's fail-safe numbers ($\alpha = 0.05$) in each analysis to determine the number of non-significant results required to change the significance of the meta-analysis (Rosenberg, 2005). Rosenberg's approach was selected over the more traditional Rosenthal method, as it better suits the meta-analytical framework by using weighted calculations and is applicable to mixed effect models (Rosenberg, 2005). A larger fail-safe number would indicate low publication bias and greater robustness of the results. To identify any disproportionate contributions from individual studies, each study was ranked by its weight (higher weights contribute relatively more to the overall mean effect size). The analyses were rerun to determine if the model significance changed when the three highest weighted effect sizes were individually removed. The assumption of normality was checked using quantile-quantile (Q-Q) plots (Zuur, 2009).

We analyzed the potential temporal effect of study length on each category of eelgrass responses (lnRR estimates) using generalized mixed effect models, assigning days-since-disturbance (log transformed) as a fixed effect and study as a random effect. Observations from all time points were included in this analysis.

3. Results

The initial literature search produced 1404 publications, 31 of which fit the criteria outlined above. These 31 papers included 125 studies that could be analyzed separately, based on different eelgrass response metrics (Table 1, Table A.1). Here we report results as lnRR, such that positive values indicate higher values of the response variable (e.g., eelgrass density) in the experimental (bivalve aquaculture) treatments relative to the control treatments (eelgrass beds). Negative lnRR values indicate the experimental treatments had lower values than the control treatments, and values close to zero indicate little or no effect.

Eelgrass response to on-bottom culture (no associated gear or predator exclusion devices) and off-bottom culture (e.g., longline and suspended bag) varied by response metric (Fig. 1, Table 2). On-bottom culture showed a 25% increase in eelgrass growth ($p < 0.05$), a 39% increase in reproduction ($p < 0.1$), a 51% decrease in eelgrass biomass ($p < 0.1$), and a 49% decrease in density ($p < 0.1$). Off-bottom culture resulted in 45% decrease in eelgrass density ($p < 0.05$), 78% decrease in percent cover ($p < 0.05$), and 61% decrease in reproduction ($p < 0.05$). The remaining responses of eelgrass metrics to on- and off-bottom culture were not significantly different from zero or lacked adequate sample sizes for analysis.

Broad regional differences explain some of the trends in on-bottom/off-bottom culture – eelgrass interactions (Fig. 2, Table 2). Sample sizes restricted our regional separation into two groups: the US west coast (the Pacific coasts of Washington, Oregon and California states, Puget Sound, WA, and one study from British Columbia, Canada), and the Global group (excluding the US west coast studies) (Table A.1). The US west sub-region and the remaining global region differed in two metrics. Off-bottom culture on the US west coast decreased eelgrass density by 50% ($p < 0.05$), relative to a neutral interaction in the global group, and on-bottom culture had a neutral effect on reproduction, relative to a 50% increase in the global group ($p < 0.1$).

Specific aquaculture gear types explained significant variation of eelgrass responses within on- and off-bottom categories, however, our analysis categories were again restricted by sample size (Fig. 3, Table 2). The lack of gear (i.e. no predator exclusion devices) used in on-bottom culture had a neutral effect on all eelgrass categories except density (45% decrease, $p < 0.05$) and growth (25% increase, $p < 0.05$). Longline gear also had a negative effect on eelgrass density (44% decrease, $p < 0.05$). Suspended bag gear type had a neutral effect across all eelgrass response categories ($p > 0.1$).

Manual harvesting methods (hand, hand blade, and hoe) had less initial impact on eelgrass density than mechanical methods (digging, dredging, sediment liquefaction, and dragging) (χ^2 (1, N = 65) = 5.6, $p < 0.01$, Fig. 4). Mechanical harvesting resulted in average initial decrease in eelgrass density lnRR of -5.18 (99% decrease), while manual harvesting resulting in an average initial decrease of -0.84 (57% decrease). Eelgrass density had a positive slope in both categories implying post-harvest recovery (Fig. 4) .

4. Sensitivity analysis

There were no changes in model significance when the three highest weighted studies were individually removed from each analysis (on/off bottom, on/off bottom and region, and gear type), showing no disproportionate influence of a specific study. The Rosenberg fail-safe numbers were relatively high for eelgrass density and growth (N = 3783, N = 1275, respectively), and moderate for eelgrass biomass (N = 282), percent cover (N = 112), reproduction (N = 260), and structure (N = 203).

All six eelgrass response metrics significantly varied with the length of time between the initial time of shellfish presence and sampling time

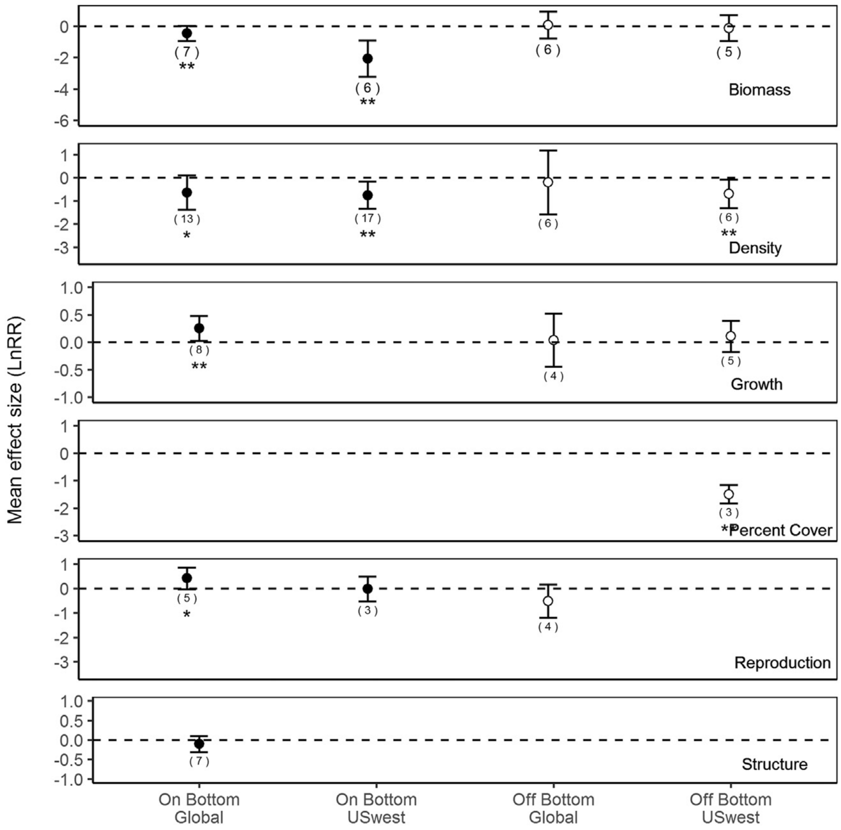

Fig. 2. The mean effect (95% confidence intervals) of on-bottom (black circles) and off-bottom (white circles) bivalve aquaculture on the US west coast relative to the Global region (all other studies not in the US west coast group), on separate analyses of eelgrass biomass, density, growth, percent cover, reproduction, and structure. Sample size is in parenthesis and significance is indicated by * (< 0.1) or ** (< 0.05). The zero dashed line indicates no effect.

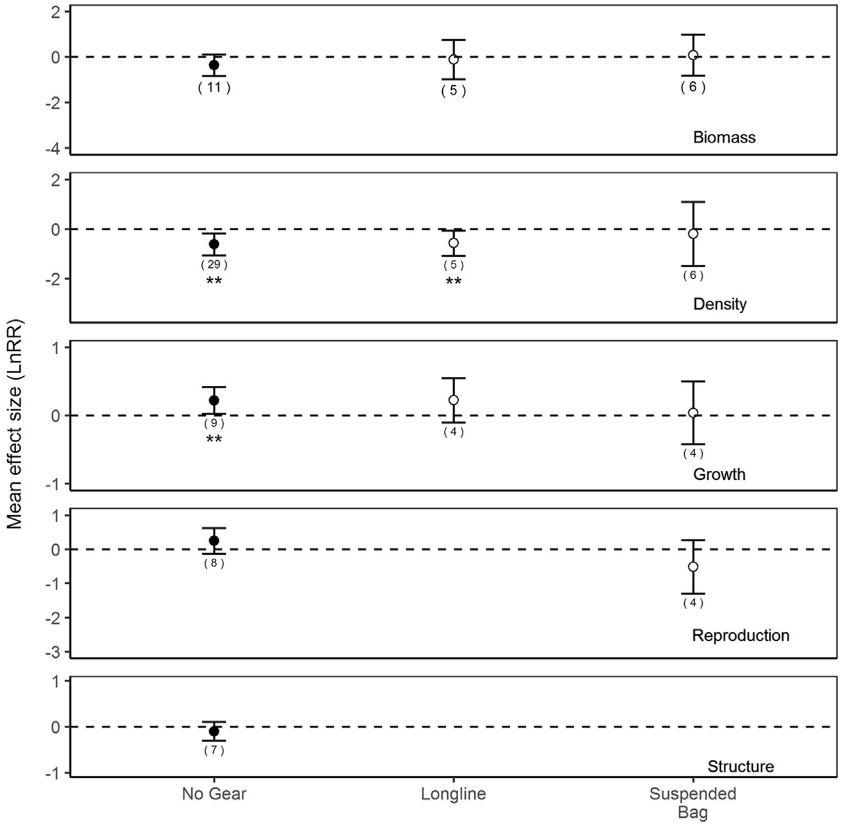

Fig. 3. The mean effect (95% confidence intervals) of no gear (on-bottom, black circle) relative to longline and suspended bag gear types (off-bottom, white circle) on separate analyses of eelgrass biomass, density, growth, percent cover, reproduction, and structure. Sample size is in parenthesis and significance is indicated by * (< 0.1) or ** (< 0.05). The zero dashed line indicates no effect.

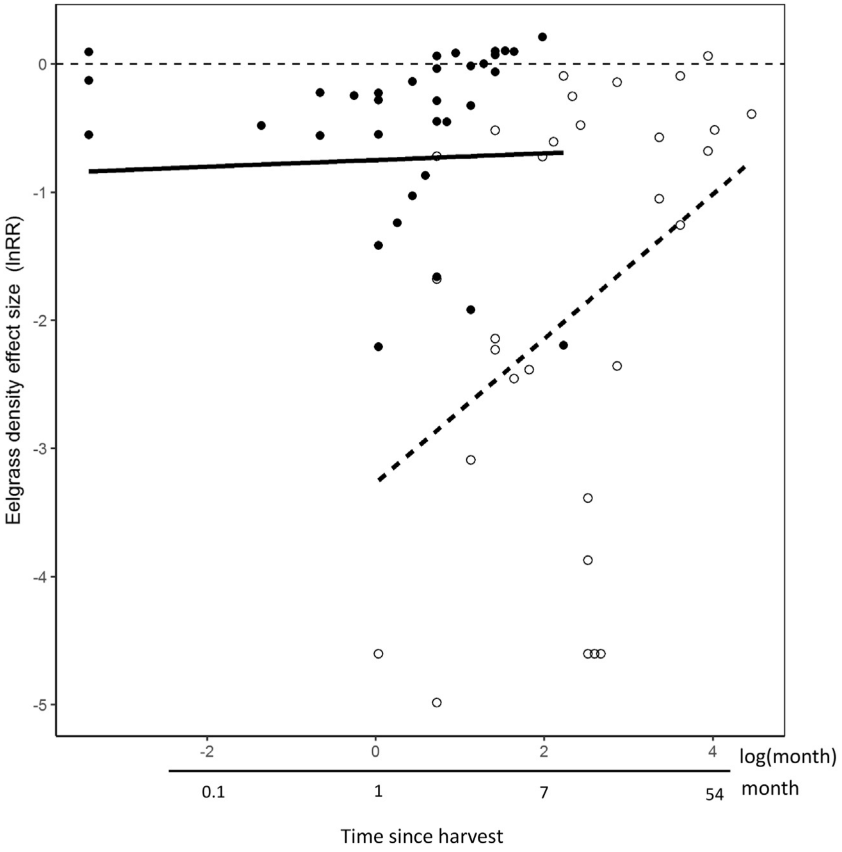

Fig. 4. Changes in eelgrass density (lnRR) in time (log month) after bivalve harvest disturbance. Harvest methods are divided into manual (closed circles and solid line: hand, hand blade, hoe, rake) and mechanical (open circles and dashed line: dragging, dredging, digging, and sediment liquefication).

Table 2

Summary of significant ($p < 0.1$) percent changes in eelgrass response metrics due to presence of shellfish aquaculture (as shown in Figs. 1–3). Eelgrass metrics increased (blue), decreased (orange), had no significant change (yellow) or had inadequate sample size for analysis (NA; gray).

	% Change in Eelgrass Response Metric					
Grow-out method & Region	Density	Percent Cover	Growth	Biomass	Structure	Reproduction
On Bottom						
All Regions	-48.91	NA	24.83	-50.60		39.46
No Gear	-46.20	NA	24.65			
Global	-47.20	NA	28.63	-36.57		51.66
US West	-53.11	NA	NA	-87.31	NA	
Off Bottom						
All Regions	-45.12	-77.58			NA	-60.66
Longline	-43.67	NA			NA	NA
Suspended Bag		NA			NA	
Global		NA			NA	
US West	-50.17	-77.58			NA	NA

($p < 0.001$; Table A.2). Eelgrass growth and structure lnRR values increased over time and eelgrass density, biomass, reproduction, and percent cover lnRR values decreased over time. These trends do not speak to lnRR magnitude, meaning eelgrass density could have a mean positive lnRR response but still have negative slope over time, or vice versa. All regions and gear types were grouped together to obtain adequate sample size, acknowledging that variation exists within these groups as described above. We did not use these temporal relationships to detrend our original lnRR data, as sample size was too limited across eelgrass metrics, gear types, and time steps to develop robust relationships.

5. Discussion

Eelgrass has variable responses to shellfish aquaculture depending on the characteristic of eelgrass being measured, the use of on-bottom or off-bottom grow-out methods, geographic region, and gear type. Off-bottom grow-out methods negatively affected eelgrass density, percent cover, and reproduction. On-bottom grow-out methods decreased eelgrass density and biomass, and increased eelgrass growth and reproduction. Varying eelgrass responses to on-bottom versus off-bottom shellfish culture suggest alternative mechanisms underlying these relationships. Differences between the US west coast and the remaining Global group suggest that regional environmental conditions or aquaculture practices can influence these cultured bivalve-eelgrass relationships. Neutral trends in eelgrass response metrics could reflect a neutral impact or be due to averaging across many environmental conditions and bivalve densities, potentially masking locally significant positive or negative trends. The temporal nature of these impacts are not analyzed in this study (except for density response to harvest), however shellfish cultivation has coexisted with eelgrass for 100s of years in many locations implying these impacts are not permanent.

Bivalves and eelgrass have the potential to compete spatially at high shellfish densities. On-bottom culture with no predator exclusion gear resulted in a decrease in eelgrass density, an increase in growth, and neutral effects on biomass, reproduction, and structure. A decrease in eelgrass density from on–bottom culture supports the space competition hypothesis (Tallis et al., 2009; Wagner et al., 2012) and the inverse relationship that can exist between eelgrass density and biomass (Yang et al., 2013). In conditions where shellfish are competing with eelgrass for space but not for other resources (e.g., light), decreased eelgrass density can result in competitive release of remaining eelgrass shoots and enhance growth (Olesen and Sand-Jensen, 1994). Site-specific studies have observed increases in shoot length and clonal branching, which may be masked by the broad grouping, and neutral results of our eelgrass reproduction and structure response categories (Kelly and Volpe, 2007; Vinther et al., 2008; Tallis et al., 2009; Ruesink and Rowell, 2012).

Off-bottom shellfish aquaculture (e.g., longlines and suspended bags) has the potential to negatively impact eelgrass through light limitation, improve the eelgrass environment via sediment stabilization and wave attenuation, or have a neutral effect on the eelgrass bed (as reviewed in Dumbauld et al., 2009, Forrest et al., 2009). The meta-analysis supported mean negative (eelgrass density, percent cover and reproduction) and neutral (biomass and growth) effects of off-bottom culture on eelgrass response metrics. The negative/neutral eelgrass responses to suspended culture lends weight to the light attenuation mechanism of interaction over the benefits of habitat modification. Site-specific suspended culture studies have documented neutral and decreased eelgrass densities and percent cover, while increased spacing of gear can reduce negative effects (Everett et al., 1995; Crawford et al., 2003; Rumrill and Poulton, 2004; Wisehart et al., 2007; Skinner et al., 2014). The lack of consensus on off-bottom-eelgrass interactions in the literature suggests a more complex interaction, reflecting multiple mechanisms, environmental conditions, and gear spacing and design (Rumrill and Poulton, 2004; Dumbauld et al., 2009). Eelgrass reproductive efforts tends to increase when eelgrass is disturbed, including asexual branching to fill gaps and sexual seed production for rapid recruitment after major disturbances (Cabaco and Santos, 2012; Ruesink et al., 2012; Yang et al., 2013; Thom et al., 2014). Off-bottom culture correlated with a negative response in eelgrass reproduction in this study, at odds with the trends found in individual studies described below. The unexpected result could be due to the averaging of asexual and sexual reproductive metrics across multiple types and levels of disturbance (within the off-bottom culture category) or the time frame of measurements after disturbance. Finally, habitat modification may not have been supported in this study due to our requirement that studies have an eelgrass reference site present nearby, thus not allowing for the possibility that eelgrass could recruit into shellfish beds where it had not existed previously.

Regional differences in growing practices and environmental conditions can influence the response of eelgrass to bivalve aquaculture. Along the US west coast, off-bottom culture negatively influenced eelgrass density, whereas this effect was neutral elsewhere, and on-bottom culture had a neutral effect on reproduction on the US west coast while this effect was positive elsewhere. The difference in eelgrass density response is partially explained by an increased representation of longline studies in the US west coast region. The meta-analysis results show longlines negatively impacted eelgrass density, whereas suspended bag (more represented in the global region) had a neutral effect on eelgrass density. The difference in eelgrass' reproductive response could indicate a greater disturbance caused by on-bottom culture in the global region relative to the US west coast (Cabaco and Santos, 2012; Yang et al., 2013; Thom et al., 2014). Averaging across a variety of environmental conditions in the Global region may contribute to the larger confidence intervals and neutral responses of eelgrass biomass, density, reproduction, and growth to off-bottom culture. The global studies included in our meta-analysis range in turbidity, temperature, water levels, nutrient levels, salinity, light limitations, and *Zostera* spp. which can influence growth, biomass, and reproduction (Yang et al., 2013; Thom et al., 2014; Hitchcock et al., 2017). Conversely, spring-summer environmental conditions in estuaries along the US west coast (NE Pacific Ocean) are characterized by relatively lower temperature, nutrient-rich, higher salinity, upwelled water from the California Current (Hickey and Banas, 2003).

Bivalve harvest method and intensity influence the initial impact and final recovery time of eelgrass density. Mechanical harvest practices (e.g., dredging) had the largest initial impact and required the longest time for recovery, potentially due to the removal or destruction of above and below ground eelgrass biomass. Longer return times for mechanical methods could also be explained by their potential for use in harvesting of larger sites, resulting in a larger area to be repopulated from an intact eelgrass source population that is farther away. Manual methods of harvest can be more spatially targeted, have less impact on the eelgrass rhizomes beneath the surface, and result in a faster recovery time (Cabaco et al., 2005; Wootton and Keough, 2016). Manual harvesting would imply a smaller area of disturbance, leaving intact eelgrass closer to the disturbed area (e.g., gap edges) and more quickly able to recolonize (Ruesink et al., 2012). Eelgrass reproductive effort can also influence recovery time, varying between clonal branching (asexual) and seed production (sexual) (e.g., Plus et al., 2003; Boese et al., 2009). Disturbance type, intensity, and frequency can influence the resilience of intertidal communities, but return time might not be the most appropriate metric for monitoring the health of eelgrass habitat after a disturbance (Short and Wyllie-Echeverria, 1996). Eelgrass recovery time assumes a stable state ecosystem as the point of return and not a dynamic equilibrium with cumulative stressors and disturbances (Gunderson, 2000). In the latter context, which better suits the marine environment, the resilience and persistence of an eelgrass bed may be better measured by its retention of ecological services and function (Thom et al., 2012). Eelgrass density has established relationships with certain ecological functions, including ability to trap sediment, primary productivity and contribution to the detrital food web (Koch, 2001; Duarte et al., 2010). There is less consensus, and more species-specific differences, on the relationships between eelgrass density and habitat value for fish, invertebrate, and infaunal communities (Turner et al., 1999; Bostrom and Bonsdorff, 2000; Blackmon et al., 2006; Hosack et al., 2006; Hirst and Attrill, 2008; Semmens, 2008; Dumbauld et al., 2015; Gross et al., 2018).

Understanding the trends in eelgrass responses to bivalve aquaculture is an important step in sustainably managing bivalve aquaculture. The benefits of a meta-analysis include the discovery of general trends that outweigh the underlying variability present in individual studies. While these meta-analyses results are not intended to supersede regionally specific studies, they may be helpful in informing best practices to locations where individual studies are not yet available. Some sources of variability that we could not account for in these analyses included the spatial and temporal scale of disturbance, for example, the size of aquaculture plots in each study and whether individual sites were repeatedly planted and harvested or were being disturbed for the first time. Accounting for study length (time since shellfish planting) has the potential to change some of these results (e.g., Dumbauld and McCoy, 2015). In addition, a greater number of studies from different regions would enable us to examine the potential influence of ocean conditions on the responses of eelgrass metrics to shellfish aquaculture.

Shellfish have been farmed in and around eelgrass for over a century and it is clear that shellfish aquaculture does not preclude eelgrass. However, as demonstrated by our study, there can be temporary changes in certain eelgrass characteristics. To account for these potential changes on the US west coast, resource managers follow avoid, minimize, and mitigate protocols outlined in federal and state management documents (e.g., NOAA Fisheries West Coast Region, 2014, US Army Corps of Engineers Seattle District, 2015). These documents currently recommend that any new farms avoid impacts to eelgrass by placing a buffer around eelgrass, or including an unvegetated perimeter in the eelgrass bed definition, and avoid working in those areas. If avoidance is not possible, such as on existing farms, impacts should be minimized (e.g., minimizing the number of workers or boats used in areas with eelgrass). If disturbance cannot be avoided or minimized then mitigation is recommended. The US west coast data analyzed in this meta-analysis were mostly collected from farms working within this management context. Best management practices continue to evolves. For example, there is a condition for Endangered Species Act coverage in Washington, USA, that oyster longlines and flipbags must be spaced laterally at 10 feet intervals in fallow areas that have been colonized by eelgrass to minimize potential impacts of shading (NOAA Fisheries 2016). Additional topics that would inform this discussion include spatial management, landscape perspectives, and connecting changes in eelgrass response metrics (e.g., density) to ecological function.

Acknowledgements

This research was supported by NOAA Fisheries Office of Aquaculture, U.S.A. (Silver Spring, MD) and NOAA Fisheries West Coast Region, U.S.A. (Seattle, WA). We thank our colleagues who provided insight and expertise that greatly assisted the research, especially Eric Buhle, Kurt Fresh, Zach Hughes, Peter Kiffney, P. Sean McDonald, Michelle McMullin, Mike Rust, Korie Schaeffer, Jennifer Ruesink, Jonathan Reum, Jameal Samhouri, Ole Shelton, Adrian Stier, Karl Veggerby, and Greg Williams.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.aquaculture.2018.08.046.

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Fisheries and Oceans Canada
Pêches et Océans Canada
Ecosystems and Oceans Science
Sciences des écosystèmes et des océans

Canadian Science Advisory Secretariat (CSAS)

Research Document 2015/002

Gulf Region

Modelling carrying capacity of bivalve aquaculture: a review of definitions and methods

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July 2015

Foreword

This series documents the scientific basis for the evaluation of aquatic resources and ecosystems in Canada. As such, it addresses the issues of the day in the time frames required and the documents it contains are not intended as definitive statements on the subjects addressed but rather as progress reports on ongoing investigations.

Research documents are produced in the official language in which they are provided to the Secretariat.

Published by:

Fisheries and Oceans Canada
Canadian Science Advisory Secretariat
200 Kent Street
Ottawa ON K1A 0E6

http://www.dfo-mpo.gc.ca/csas-sccs/
csas-sccs@dfo-mpo.gc.ca

ISSN 1919-5044

Correct citation for this publication:

Filgueira, R., Comeau, L.A., and Guyondet, T. 2015. Modelling carrying capacity of bivalve aquaculture: a review of definitions and methods. DFO Can. Sci. Advis. Sec. Res. Doc. 2015/002. v + 31 p.

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ABSTRACT

The need to develop sustainable activities in the coastal zone that are compatible with other uses is key for minimizing controversy among stakeholders and consequently for marine spatial planning. In aquaculture, carrying capacity (CC) has traditionally been regarded as a useful concept for maximizing stocking biomass and profitability at the farm scale. Over the past decades the concept itself and its level of complexity have substantially evolved, prompting the creation of different CC components: physical, production, ecological, social and governance. These components provide a holistic approach to the analysis of aquaculture and its interaction with the ecosystem, economy and society. With respect to ecological and production CC, a series of indices and complex mathematical models have been developed. Nevertheless, there is still no consensus regarding criteria and thresholds, a situation that often limits the application of these new tools to management. In this document, we review the current understanding of CC for bivalve aquaculture, with emphasis on the modelling techniques that are available for inferring ecological and production CC.

Modélisation de la capacité de charge de l'aquaculture des bivalves : un examen des définitions et des méthodes

RÉSUMÉ

Il est essentiel de planifier des activités durables qui sont compatibles aux autres utilisations dans la zone côtière de manière à minimiser toute controverse parmi les intervenants, permettant ainsi la planification spatiale maritime. En aquaculture, la capacité de charge a traditionnellement été un concept utile afin de maximiser la biomasse en production et la rentabilité à l'échelle de l'exploitation. Toutefois, au cours des dernières décennies, le concept lui-même et son niveau de complexité ont considérablement évolué, entraînant du même coup la création de différents aspects de la capacité de charge : physique, production, écologique, social et gouvernance. Ces aspects offrent une approche holistique à l'analyse de l'aquaculture et de ses interactions avec l'écosystème, l'économie et la société. En ce qui concerne les aspects écologique et relatif à la production de la capacité de charge, une série d'indices et de modèles mathématiques complexes ont été mis au point. Pourtant, il n'y a pas encore de consensus en ce qui a trait aux critères et aux seuils; cette situation limite souvent l'application de ces nouveaux outils dans un contexte de gestion de l'aquacuture. Dans le présent document, nous passons en revue la compréhension actuelle de la capacité de charge par rapport à l'aquaculture des bivalves en mettant l'accent sur les techniques de modélisation disponibles afin d'inférer les aspects écologique et relatif à la production de la capacité de charge.

INTRODUCTION

The rapid growth of aquaculture at a worldwide scale (Costa-Pierce 2010) has generated concerns about the impacts of the activity not only on local environments but also at the social scale, as aquaculture competes for space and resources with traditional users of land, water and coasts (Byron and Costa-Pierce 2013). Carrying capacity (CC) is one of the most useful concepts to assess the development of shellfish mariculture, but remains contentious due to the lack of a clear and concise definition (McKindsey et al. 2006). The importance of CC is reflected in the growing number of reviews on the subject (McKindsey et al. 2006; Grant and Filgueira 2011; Byron and Costa-Pierce 2013; Ferreira et al. 2013; McKindsey 2013). In this document, we provide an overview of the current understanding of CC, particularly ecological and production CC, by focusing on the methodologies that are presently available for their inference. We present this information using the following structure:

- Bivalve aquaculture and the environment
    - Pelagic environment
    - Benthic environment
    - Habitat modification/creation
- Definition of carrying capacity
- Inferring carrying capacity: indices, models and thresholds of sustainability
    - Indices
    - Farm models
    - Spatial models
    - Food web models
- Conclusions

BIVALVE AQUACULTURE AND THE ENVIRONMENT

Bivalves act as ecosystem engineers or foundation species by influencing habitat and resources available for other species (Jones et al. 1997). This section has been divided into three sub-sections. The first addresses the direct role of bivalves on the pelagic environment with emphasis on bivalve-phytoplankton trophic interaction as well as nutrient cycling and particle dynamics. The second sub-section concerns the effect of bivalves on the benthic environment with emphasis on potential modification of the characteristics of the seafloor and associated communities. The third sub-section focuses on the potential consequences of culturing bivalves on habitat. In each section we have also discussed the most significant impacts that bivalve aquaculture may exert on the environment, which have been identified by McKindsey (2013) as: food depletion in the water column; enhanced localized biodeposition and the concomitant alteration of nutrient, oxygen fluxes, and community composition; and transfer of diseases and hitchhiking species. Although we focus on the major potential impacts, the different stages of the aquaculture process can lead to different impacts (Table 1). In addition to McKindsey (2013), a series of reviews with focus on the potential environmental impacts of bivalve aquaculture have been recently published. Dumbauld et al. (2009); Forrest et al. (2009) and McKindsey et al. (2011) provide a detailed list of potential effects and environmental interactions with clams, oysters and mussels, respectively. Cranford et al. (2012) also provides a general overview for bivalve aquaculture without emphasis on any particular species, and summarizes a series of indicators of benthic and pelagic impacts.

PELAGIC ENVIRONMENT

The cultivation of bivalves at high densities can exert a significant effect on flows of matter and energy in coastal marine ecosystems (Figure 1.1 in Cranford et al. 2006) (Dowd 2003; Cranford et al. 2012). The most obvious effect of introducing a large biomass of filter-feeders is the interaction with phytoplankton populations. Bivalve filtration activity may exert a top-down control of phytoplankton populations (Dame 1996; Dame and Prins 1998; Newell 2004; Petersen et al. 2008; Huang et al. 2008), to the extent that if bivalve culture consumes the phytoplankton faster than it can be replaced by advection or local production, the resulting depletion can compromise bivalve performance itself (Bacher et al. 2003; Strohmeier et al. 2005; Ferreira et al. 2007; Duarte et al. 2008). The concept of “depletion” though commonly used in the bivalve literature could be mistakenly understood as having a negative connotation. The term should not be interpreted as a near-total absence of phytoplankton cells. Instead the term simply refers to a reduction of phytoplankton concentration attributed to bivalve grazing, a phenomenon that naturally occurs over dense bivalve beds (Dame et al. 1991; Dolmer 2000). In addition to reducing phytoplankton concentrations, filtration activity can alter the structure of phytoplankton communities. For example, in light-limited systems, filtration can decrease turbidity and facilitate a shift towards faster growing algal species (Prins et al. 1995). Also, the progressive increase of bivalve retention efficiency from small to large particles (Strohmeier et al. 2012) can result in increased picophytoplankton abundance (Olsson et al. 1992; Vaquer et al. 1996; Dupuy et al. 2000; Souchu et al. 2001; Cranford et al. 2008; Smaal et al. 2013; Froján et al. 2014). Although most research on bivalve feeding behaviour has been focused on the interaction between bivalves and phytoplankton, there is support in the literature that detritus can also be important in bivalve nutrition (Hawkins et al. 2002; Navarro et al. 2009). Consequently, selective retention efficiency could lead to an alteration of the structure of the phytoplankton community, as well as an alteration of detritus size spectra, both of which may alter the available seston for other consumers (Prins et al. 1998; Dupuy et al. 2000; Pietros and Rice 2003; Leguerrier et al. 2004). Despite the breadth of research on bivalve feeding behaviour including feeding rate, retention efficiency, particle sorting and digestive processes, there are still major knowledge gaps in our understanding of what bivalves eat. This also includes their interactions with marine aggregates, commonly known as marine snow, which can be ingested by bivalves (Newell et al. 2005) and may play a key role in benthic-pelagic coupling.

During the feeding process, phytoplankton and particulate organic matter are consolidated into pseudofaeces (uningested) and faeces (unassimilated). Remineralization of nutrients in pseudofaeces and faeces in the water column, and particularly in the benthos (Grant et al. 1995; Newell 2004; Carlsson et al. 2010; Jansen et al. 2012), as well as bivalve ammonia excretion can be directly used by phytoplankton (Dame et al. 1991, Smaal and Prins 1993, Prins et al. 1998; Mazouni 2004; Asmus and Asmus 2005; Sará 2007; Cranford et al. 2007). The ecological significance of nutrient regeneration consists of a relaxation of nutrient limitation for phytoplankton, which may result in enhanced primary production at the local scale (Smaal 1991; Prins et al. 1995; 1998; Pietros and Rice 2003), a process that is more significant in nutrient-limited environments. Therefore, cultured shellfish can also exert a bottom-up nutrient control on phytoplankton populations (Ogilvie et al. 2000; Cranford et al. 2007; Trottet et al. 2008; Froján et al. 2014), which is considered to be an important feedback of bivalves on their own food source (Smaal et al. 2001). These processes have been demonstrated in mesocosm studies with the clam *Mercenaria mercenaria*, which showed that a relatively low abundance of clams can double primary production and alter the community structure of phytoplankton (Doering and Oviatt 1986; Doering et al. 1989). However, this positive feedback on phytoplankton populations is limited and increasing bivalve biomass will reach a point at which grazing on phytoplankton begins to reduce primary production (Smaal et al. 2013).

According to this, the most studied pelagic impact of bivalve farming is related to the top-down pressure that bivalves can exert on phytoplankton (see above). Top-down pressure can reduce phytoplankton concentrations to below natural levels, a situation that is commonly referred to as “phytoplankton or seston depletion”. Percent depletion within bivalve farms can range from nearly undetectable levels to 80% (Petersen et al. 2008; Duarte et al. 2008; Cranford et al. 2014). Seston depletion can be enhanced by the filter-feeding activity of fouling organisms and other biota associated with bivalve aquaculture (Mazouni et al. 2001; Mazouni 2004; Decottignies et al. 2007). A more intense phytoplankton depletion is expected at the local scale in the vicinity of farms (Gibbs 2007; Grant et al. 2007; Dumbauld et al. 2009; Lin et al. 2009) but at a lower intensity, effects may also extend to a larger spatial scale (Prins et al. 1998; Cerco and Noel 2007; Coen et al. 2007; Grant et al. 2008; Lin et al. 2009; Filgueira and Grant 2009; Filgueira et al. 2014a). Intense depletion is expected at shallow sites characterized by long water residence times and high bivalve densities, such as densities found within three-dimensional culture systems (Rodhouse and Roden 1987; Heral 1993; Comeau et al. 2008). Depletion may lead to increased competition for phytoplankton, which may ultimately have a significant effect on bivalve growth rate and performance (Filgueira et al. 2013; 2014a). Depletion can also negatively affect other grazers in the surrounding ecosystem, such as wild populations of filter feeders (van Strahlen and Dijkema 1994), including zooplankton (Gibbs 2007). Consequently, farmed bivalves can redirect energy away from zooplankton, replacing the ecological role of zooplankton in the ecosystem by grazing on phytoplankton (Jiang and Gibbs 2005; Gibbs 2007). Bivalves can also directly predate on zooplankton populations, although this pathway of effect has not been extensively documented (Davenport et al. 2000; Trottet et al. 2008). Perturbations at the zooplankton trophic level can alter the energy transfer to higher trophic levels because zooplankton is preyed upon by small fishes, especially fish larvae (Gibbs 2007), but cultured bivalves are generally not preyed upon by small fishes (Gibbs 2007).

BENTHIC ENVIRONMENT

Faeces and pseudofaeces sink to the seafloor (Newell 2004; Dame 1996) and redirect part of the pelagic energy flow towards benthic food webs (Leguerrier et al. 2004; Cranford et al. 2007; Lin et al. 2009) where they can be further recycled (Cloern 1982; Dumbauld et al. 2009). These vertical fluxes of wastes (generally referred to as biodeposits) can potentially increase the accumulation of organic material in the vicinity of the aquaculture site (McKindsey et al. 2006), which can alter benthic habitat and communities. Although, the main source of organic loading is in the form of biodeposits, the fall-off of farmed bivalves and associated fouling can also constitute a significant contribution (McKindsey et al. 2011). Biodeposits are generally organic-rich, fine particles and consequently the seabed can become organically enriched and fine-textured relative to surrounding areas. The effects of the biodeposits on the benthos depend on local environmental characteristics and the intensity of farming activity (Giles et al. 2009).

The production rate of biodeposits is directly dependent on the stocking density of a cultured population (Forrest et al. 2009; Dumbauld et al. 2009; McKindsey et al. 2011) and the organisms associated with the farmed population (Stenton-Dozey et al. 2001; Giles et al. 2006; McKindsey et al. 2009). Other important factors that determine how biodeposits are deliverd to the benthos are their sinking velocity, which is dependent on a bivalve’s diet (Chamberlain 2002; Giles 2006), and the physical flushing of the water in the embayment. The sinking velocity and physical flushing will determine the dispersion of the biodeposits (Pearson and Black 2001; Grant et al. 2005). Note that physical flushing may also be affected by the farming structures themselves, since they can alter hydrodynamics (Plew et al. 2005; Petersen et al. 2008;

Stevens et al. 2008; Strohmeier et al. 2008; Comeau et al. 2014). In general, the deposition of biodeposits of bivalves and associated fauna is restricted to the area directly below the farm and its proximal vicinity, where sedimentation rates can be significantly enhanced compared to reference sites outside the farm (Tenore et al. 1982; Hatcher et al. 1994; Hartstein and Stevens 2005). However there are potentially serious issues associated with sampling designs and reference sites. Recently, Guyondet et al. (2014) stated that the increase in sedimentation is confined to the central zone of the farmed areas while a reduction is expected outside the farms. This highlights the serious issues associated with sampling designs and establishing reference sites.

The ultimate effects of the organic loading depend on :

1) the characteristics of the biodeposits (Giles and Pilditch 2006; Carlsson et al. 2010), which can greatly vary both spatially and temporally (Bayne et al. 1993);
2) local sediment characteristics such as grain size, temperature, dissolved oxygen, presence/absence of seagrass, infauna, etc.; but also
3) the coupling with the water column, e.g. oxygen fluxes.

Together, these variables ultimately determine the assimilative capacity of the sediments (Kusuki 1981; Souchu et al. 2001; Mitchell 2006). In general, the increase in organic loading can alter benthic sediment geochemistry (Dalhlbäck and Gunnarsson 1981; Mattsson and Lindén 1983). The organic matter can be consumed by benthic feeders, mitigating the effects of organic loading (Valdemarsen et al. 2010), but it can also be decomposed by microbes following a series of oxidant reductions (Figure 2 in McKindsey et al. 2011). When the decay of biodeposits consumes oxygen at a rate greater than that of renewal by water exchange, this can create anoxic conditions and induces sulfate reduction near the surface of the sediment (McKindsey et al. 2011). Concomitant with the increase in organic matter, an increase in anoxic conditions and total free sulphides as well as a reduction in redox potential may occur beneath a bivalve farm (Tenore et al. 1982; Hargrave et al. 2008). Consequently, fluxes of ammonium, nitrate/nitrite, phosphate and silicate can be enhanced below bivalve farms (Giles et al. 2006; Carlsson et al. 2009; Dumbauld et al. 2009; Alonso-Pérez et al. 2010). Organic loading can create, in a worst case scenario, locally anoxic sediments (Forrest and Creese 2006; Nizzoli et al. 2006; Mesnage et al. 2007; Richard et al. 2007; Hargrave et al. 2008) and increasing local sulfide concentrations (Holmer et al. 2005; Vinther and Holmer 2008; Vinther et al. 2008).

The chemical changes as well as the physical changes in the sediment characteristics potentially enhance differences between benthic communities within and outside of culture sites (McKindsey et al. 2011). These changes can significantly affect habitat characteristics and ultimately infaunal communities (Mirto et al. 2000; Christensen et al. 2003; Beadman et al. 2004; Callier et al. 2007; Murray et al. 2007). The response of benthic communities follows the classical conceptual model of succession proposed by Pearson and Rosenberg (1978). These effects range from no discernible effects in the infaunal community (Crawford et al. 2003; Mitchell 2006) to significant alterations of the infaunal assemblage (Castel et al. 1989; Nugues et al. 1996; Hartstein and Rowden 2004), including intermediate situations in which moderate effects have been identified (Forrest and Creese 2006). The effects can also be significant in microbial and meiofaunal communities (Mirto et al. 2000). Another key aspect is related to the potential effects on bioturbators, which have a great impact on oxygen dynamics in sediments (Norkko and Shumway 2011). Gibbs (2007) suggested that benthic communities could shift from filter- to deposit-feeders. In general, the response of the macrobenthic community to the increase in organic loading follows these steps (Cranford et al. 2012):

- a decrease in species richness and an increase in the total number of individuals as a result of the high densities of a few opportunistic species;
- a general reduction in biomass for most species, although there may be an increase in total biomass corresponding to the presence of a few opportunistic species;
- a decrease in body size of the average species or individual;
- a shallowing of that portion of the sediment column occupied by infauna; and
- a shift in the relative dominance of trophic groups.

Ultimately, in a worst-case scenario, the sediment may become anoxic and dominated by microbial mats such as *Beggiatoa* spp (McKindsey et al. 2011; Norkko and Shumway 2011). *Beggiatoa* spp may also be present in early stages of benthic degradation.

HABITAT MODIFICATION/CREATION

In addition to the modification of benthic habitat via organic loading (see above), bivalve aquaculture can exert other modifications in both benthic and pelagic habitat. The increase in light penetration (Cerco and Noel 2007; Schröder et al. 2014) and/or sediment nutrient enrichment (Reusch and Williams 1998; Peterson and Heck 1999; 2001) promoted by cultured bivalves can enhance the productivity of seagrass in shallow coastal ecosystems (Peterson and Heck 2001; Carroll et al. 2008). For example, it is predicted that the increase of water clarity from bivalve aquaculture and/or oyster restoration may lead to an increased biomass of submerged aquatic vegetation (Newell and Koch 2004; Cerco and Noel 2007; Wall et al. 2008). A recent study conducted in Atlantic Canada showed a positive relationship between farmed oyster biomass and eelgrass (*Zostera marina*) biomass (Andrea Locke, personal communication). In addition to the direct effect on benthic habitat caused by the proliferation of submerged aquatic vegetation, a reduction of phytoplankton production in the water column is expected, due to the direct competition for nutrients (Souchu et al. 2001; Newell 2004; Porter et al. 2004).

Another effect of bivalve aquaculture on habitat availability is related to the deployment of three-dimensional physical structures in the ocean (Costa-Pierce and Bridger 2002; Olin 2002, Davenport et al. 2003; McKindsey et al. 2011). The physical structure can provide refuge from predators and/or food sources (see McKindsey et al. 2011), which can attract new species (Gibbs 2004; Jiang and Gibbs 2005; Clynick et al. 2008). For example, Tenore and González (1976) found that mussel rafts in Ría de Arousa (Spain) provided habitat and food resources that enhanced secondary productivity. Tallman and Forrester (2007) show that oyster grow-out cages provide quality habitat for fishes associated with hard-bottom habitats. Physical structures such as farms or docks can also create shaded areas (Everett et al. 1995; Burdick and Short 1999; Thom et al. 2005), which can negatively impact photosynthetic organisms, such as seagrass (Skinner et al. 2013; 2014). The structures can alter hydrodynamics (Plew et al. 2005; Petersen et al. 2008; Stevens et al. 2008; Strohmeier et al. 2008; Comeau et al. 2014), causing erosion or deposition, depending on local characteristics (Dumbauld et al. 2009; McKindsey et al. 2011). These structures can also generate attachment points for epiphytic species, including macroalgae (Dumbauld et al. 2009) but also invasive species, including tunicate ascidians such as *Ciona intestinalis* and *Styela clava*. The latter species cause serious problems for aquaculture operations worldwide by increasing operational costs and/or causing direct competition for food with cultured bivalves (Karayucel 1997; Uribe and Etchepare 2002; Carver et al. 2003; Lambert 2007; McKindsey et al. 2007).

Marine species have been transported for aquaculture purposes since at least the 1700s (Padilla et al. 2011), which has facilitated the spread of non-indigenous species, from macroanimal and macroalgal to microparasites, including toxic microalgae and diseases (McKindsey et al. 2007; Forrest et al. 2009; Padilla et al. 2011). Ruesink et al. (2005) have estimated that more than 40% of non-indigenous marine species in Europe, the western United States, and the North Sea may have been introduced by oyster aquaculture. These species can be harmful to humans, other species, and the cultured species themselves (McArdle et al. 1991; Lilly et al. 2002; Cohen and Zabin 2009; Hégaret et al. 2009). New guidelines for transfer protocols of non-indigenous species have likely reduced the transfer of unwanted organisms, especially at international scales (McKindsey et al. 2007). However, secondary transfers from aquaculture facilities, which act as reservoirs of non-indigenous species, to other locations at regional or local scale still constitute a pathway to new invasions (Forrest et al. 2009). This is related to the fact that aquaculture facilities promote the aggregation of living biomass in the ocean:

1) aquaculture implies by definition the culture of organisms at high densities;
2) farming structures are deployed in the ocean, which generate new habitats (refuge and food) that trigger the aggregation of pelagic species; and
3) fouling species can colonize cages, ropes and other gear, increasing the biomass density (Carver et al. 2003; Ramsay et al. 2008).

The biomass aggregates can act as reservoirs of non-indigenous species and increase the risk of disease development/transmission, which could transfer to new locations by secondary pathways such as hydrodynamics or local vessels.

DEFINITION OF CARRYING CAPACITY

Sustainability is easier to plan than to retrofit (Ferreira et al. 2010), which makes a case for the analysis of CC at the ecosystem scale (Ferreira et al. 2013). Inglis et al. (2000) and McKindsey et al. (2006) have defined four types of CC:

- Physical CC describes the area that is geographically available and physically/chemically adequate for a certain type of aquaculture. It is useful to quantify the potential area available for aquaculture but it provides little information for management/regulation.
- Production CC has been defined from different perspectives:
    1) from a biomass perspective as the maximum level of aquaculture production (Inglis 2002);
    2) from an economic perspective as the standing stock at which the annual production of the marketable cohort is maximized (Bacher et al. 1998; Smaal et al. 1998). Given the trade-offs between bivalve growth rates, market tastes, economic returns, etc., production CC is not necessarily the greatest biomass that can be farmed (McKindsey 2013); and
    3) from a trophic web perspective as the biomass that could be theoretically produced if the pelagic food web were collapsed down into a nutrient-phytoplankton-bivalve loop (Gibbs 2004).
- Ecological CC is defined as the magnitude of aquaculture production that can be supported without leading to unacceptable changes in ecological process, species, populations, or communities in the environment (Byron and Costa-Pierce 2013). While production CC focuses only on the target species and associated processes that support production, ecological CC should, in principle, consider the whole ecosystem and all the activities involved in the aquaculture process (McKindsey et al. 2006).

- Social CC can be defined as the amount of aquaculture that can be developed without adverse social impacts. The determination of social CC requires the involvement of multiple stakeholders, that is, scientists, farmers, regulators, NGOs, etc. (See Fig. 3 in Byron et al. 2011a). Communication between scientists and stakeholders is a key aspect of this process to obtain the expected outcome for sustainable management of resources and equity of all users (Byron et al. 2011a). Social CC is still in its infancy (Byron et al. 2011a) and analytical methods to estimate it are yet under development (Byron and Costa-Pierce 2013).

Recently, regulatory or governance CC has been added to this framework (Byron and Costa-Pierce 2013; Ferreira et al. 2013). Regulatory CC establishes the legal framework and communication protocols among stakeholders to ensure a smooth integration of the aquaculture activity into the coastal zone, minimizing potential conflicts. Governance is not usually limited by a lack of legal instruments but often by their adequacy and acceptance by stakeholders (Ferreira et al. 2013).

Assuming that physical CC cannot be used in the management/regulatory process and in many respects is encapsulated in the production CC (Ferreira et al. 2013), social CC is the most restrictive type of CC in Europe and North America (Figure 2 in Byron and Costa-Pierce 2013), where social opposition is at the forefront of decision-making (Ferreira et al. 2013). In other locations such as China and Southeast Asia, where food production is the paramount concern, social pressure adopts an entirely different perspective, acting as a driver for increased aquaculture for reasons of economy and food security (Ferreira et al. 2013). Consequently, constraints depend on social and political standpoints and cannot be homogenized worldwide. Nevertheless, a common framework for CC has been suggested following the three principles of the Ecosystem Approach to Aquaculture (EAA) (FAO 2008; Soto et al. 2008):

- aquaculture should be developed in the context of ecosystem functions and services with no degradation of these beyond their resilience capacity;
- aquaculture should improve human wellbeing and equity for all relevant stakeholders; and
- aquaculture should be developed in the context of (and integrated to) other relevant sectors.

The application of the EAA principles is a strategy for the integration of the activity within the wider ecosystem in such a way that it promotes sustainable development, equity, and resilience of interlinked social and ecological systems (Soto et al. 2008).

INFERRING CARRYING CAPACITY: INDICES, MODELS AND THRESHOLDS OF SUSTAINABILITY

The EAA concept can be applied following different frameworks (Walker et al. 2003; Nobre et al. 2009; Ostrom 2009; Cranford et al. 2012; McKindsey 2013) in which CC estimations play a key role. In this review we have focused on the two types of CC that can be estimated using mathematical modelling, that is, production and ecological CC (Ferreira et al. 2013). Models are powerful tools for exploring ecological CC, allowing for the study of stocks, energy fluxes and potential interactions in complex ecosystems (Dowd 2005). Models integrate time and space, which is critical for understanding ecological dynamics and therefore how natural systems provide ecosystem services (Palmer et al. 2004). In addition, scenario building ("what if" scenarios) allows the exploration of future situations where unanticipated stressors generate new risks or opportunities, and is thus an important tool for managing those changes (Nobre et al. 2010). Modelling is also among the few tools capable of assessing aquaculture sustainability while also considering the cumulative effects of additional human activities (e.g. eutrophication, climate change) and resident and invasive suspension-feeding species (Cranford et al. 2012). When modelling is combined with stakeholder input, the resulting ecological CC calculations are exceptionally powerful in the management arena (Byron et al. 2011a). In addition, models could be used to address the principles of EAA by:

- exploring the effects of aquaculture activity on the resilience of a relevant process of the ecosystem;
- providing objective scientific knowledge, which facilitates the communication among stakeholders; and
- exporting the outputs of the model to marine spatial planning processes in order to ensure aquaculture is compatible with other activities.

As stated above, production CC models can be limited to the target species and associated processes that support production, but ecological CC should in principle consider the whole ecosystem (McKindsey et al. 2006), that is, benthic and pelagic environments. Although both environments are obviously linked, the impacts on the benthos are usually restricted to the vicinity of the farm (Hargrave et al. 2008; Cranford et al. 2009; Guyondet et al. 2014). By contrast, the impacts on the pelagic environmet can reach a larger spatial scale due to effects on the base of the trophic web, that is, phytoplankton populations. The potential larger impact of bivalve aquaculture on the pelagic environment and the need to reduce model complexity as well as scientific uncertainty (FAO 2008) have been used as rationale to focus CC models on the pelagic environment. Actually, most CC modelling to date has been focused on the pelagic environment, with emphasis on food depletion, rather than on the benthic environment (McKindsey 2013). The methodological approaches to study CC from the pelagic perspective can be summarized in indices, farm models, spatial models and food web models.

INDICES

Indices based on the comparison of key oceanographic and biological processes have been used to assess the CC of bivalve aquaculture sites. These indices generally consider the bay as a homogenous system (1 box) and consequently lack spatial resolution (Grant and Filgueira 2011). However, their simplicity and the fact that the demand for data is relatively easy to fulfill, have resulted in their popular use in the literature (Incze et al. 1981; Cloern 1982; Officer et al. 1982; Carver and Mallet 1990; Smaal and Prins 1993; Heip et al. 1995; Comeau et al. 2008; Comeau 2013) as well as in certification programs such as the one by the Aquaculture Stewardship Council (ASC 2012). Such indices were used to explore production CC (e.g. Newell and Shumway 1993) as well as ecological CC (e.g. Gibbs 2007). Although there are slight variations, the common rationale of CC indices relies in comparing the energy demand of bivalve populations (based on filtration rates) and the ecosystem’s capacity to replenish these resources, which depends on advection and local production. For example, Dame and Prins (1998) proposed indices of depletion based on ratios among water residence time (RT), primary production time (PT) and bivalve clearance time (CT). Using these indices they evaluated the ecological CC of several bivalve aquaculture sites across the world. Their study has become well known, and their indices are now commonly referred to as ‘Dame Indices’. They provide a simple way to gauge the influence of bivalves on ecosystem processes, namely phytoplankton communities (Grant and Filgueira 2011).

Cranford et al. (unpublished manuscript) have calculated Dame Indices for embayments in Prince Edward Island, Québec and Nova Scotia (Table 2). The interpretation of Dame Indices could follow Gibbs (2007), who established specific thresholds for ecological and production CC. Gibbs (2007) suggested that when value of the ratio of carbon extracted by bivalves to carbon produced by phytoplankton (CT/PT in Table 2) is close to 1.0, the culture has reached production CC because the pelagic food web has collapsed down into a nutrient-phytoplankton-bivalve loop (Gibbs 2004). Smaal and Prins (1993) and ASC (2012) have also defined sustainable aquaculture (~ecological CC) in terms of the CT/PT ratio, namely for situations where internal food production, as opposed to water advected from outside the system, primarily regulates CC. They suggest that the CT/PT threshold for ecological CC should in reality be above 3 since it allows for an algal buffer stock to realize a certain level of primary production and also to take into account the occurrence of other unknown grazers in proximity to the bivalve farming operations. The lack of strong rationale for establishing this threshold was emphasized by ASC (2012), in which it is stated that the threshold is considered as a practical figure for management, monitoring and certification rather than an ecological fixed requirement.

In regards to ecological CC, Gibbs (2007) suggested that water residence time / bivalve clearance time (RT/CT) ratio values below 0.05 indicate that the culture will not be able to induce significant changes to the pelagic functioning, thereby meeting the definition of ecological CC. Note that Cranford et al. (unpublished manuscript) have calculated the inverse of that ratio, that is, CT/RT (Table 2), which based on Gibbs (2007) should be above 20 to meet ecological CC. However, the Gibbs' threshold of ecological CC (RT/CT < 0.05 or CT/RT > 20) is not based on any strong and objective ecosystem criterion. For example, it has been argued that bivalves naturally exerted a dominant effect in some coastal systems prior to the development of aquaculture. Historical (c. 1880–1910) baselines for North American oyster reefs suggest CT/RT values ≤ 1 for six of eight estuaries in the Gulf of Mexico (zu Ermgassen et al. 2013). A similar top-down control by oysters likely existed during pre-colonial times for some estuaries on the North Atlantic coast (Newell 1988; Mann et al. 2009a; 2009b; zu Ermgassen et al. 2013).

In summary, the lack of established thresholds for Dame-type indices presently limits their broad applicability. Another current drawback of these indices is their lack of spatial resolution, thus eliminating the possibility of differentiating areas within embayments. Inner and outer areas of an embayment, for example, are typically very different in terms of physical properties such as water renewal time. To address this issue, Guyondet et al. (2005; 2013) integrated a 3D circulation model into one of the Dame indices (CT/RT), thus allowing the calculation of the index in different sub-regions of an estuary. Results highlighted the spatial variability of the index, which was driven by variability in water residence time.

FARM MODELS

Production and ecological CC are strongly related (McKindsey 2013). In fact, the mathematical models that explore production and ecology can share the same core biogeochemical and hydrodynamic equations. However, farm-scale models restrict the model domain to the extent of the farm (Figure 1 in Ferreira et al. 2007), preventing the possibility of a general overview/assessment of the ecosystem. For this reason, farm-scale models are limited to production CC. Nevertheless, some modelling approaches at the farm scale can provide valuable understanding of farm- to ecosystem-scale impacts of bivalve culture (Cranford et al. 2014). All farm models are focused on the interaction bivalve-phytoplankton, since phytoplankton depletion is most obvious at this local scale (Grant et al. 2007; Duarte et al. 2008; Cranford et al. 2014). At the farm scale, phytoplankton dynamics are dominated by physical (advection) rather than biogeochemical (primary productivity) processes (Duarte et al. 2005). Consequently, farm-scale models usually include a hydrodynamic model to describe water circulation through the farm and a bioenergetic model to describe bivalve filtration and growth. The bioenergetic model is usually based on the Scope For Growth (SFG) approach (Winberg 1960) or a Dynamic Energy Budget (DEB) approach (Kooijman 2010). However, when the model is focused only on phytoplankton depletion, a simple flow equation based on average clearance rate of the bivalves is used to describe the phytoplankton-bivalve interaction. In the latter case, the model lacks the capability to predict bivalve growth (e.g. Incze et al. 1981). Farm models are generally considered useful for optimizing lease geometry and configuration (Pilditch et al. 2001; Aure et al. 2007; Rosland et al. 2011) as well as aquaculture husbandry (Duarte et al. 2008). They are especially useful when coupled to economic models, since they allow the estimation of optimal production based on economic profit (FARM™, Farm Aquaculture Resource Management; Ferreira et al. 2007).

SPATIAL MODELS

This category is the broadest among CC models. It includes a variety of approaches that share the same philosophy but with modelling schemes that vary in spatial resolution and complexity. Two main classes of spatial models can be established, box and fully-spatial models. The main difference between these approaches is the spatial resolution of the model. In a box model, the domain is divided into few large areas that are considered homogeneous. In fully-spatial models, a grid with hundreds or thousands of polygons is defined to represent the model domain. Consequently, the spatial resolution of box models is coarser than for fully-spatial models. Resolution has also a direct implication for how hydrodynamics are prescribed in each type of model. Box models need to accommodate the circulation of a large area of the ocean by using averaged values for each box. On the contrary, fully-spatial models are usually built using the grid that was used to develop the hydrodynamic model, making the coupling process between the hydrodynamic and biogeochemical models straightforward in terms of spatial resolution.

High spatial resolution is consistent with ecosystem-based management goals that involve marine spatial planning (Douvere 2008). In addition, spatial resolution can have implications for model predictions (Melbourne-Thomas et al. 2011), especially when processes are dependent on concentration (Fennel and Neumann 2004). Thus, the highest spatial resolution is always desirable, but complex fully-spatial models can require a significant amount of data in order to create and test the model. However, the ecological information available for validation of the performance of models is usually sparse in space and time (Kremer et al. 2010). While increasing model complexity attempts to bolster ecological realism, imperfect knowledge of relationships and parameters may also lead to greater scientific uncertainty (FAO 2008). Therefore, the assumption that extra detail is always beneficial can be flawed when applied at the scale and number of dimensions of end-to-end models (Fulton 2010). This implies that modelling should restrict its focus to relevant components and critical dynamics, which must be defined based on the management question to be addressed, available data, the important system features (including forcing conditions) and the appropriate scales (FAO 2008; Fulton 2010). Ferreira et al. (2013) summarized the desirable properties of a modelling system to address CC (Table 3).

Box and fully-spatial models share the same biogeochemical core, which can vary in complexity, ranging from simple nutrient-seston-bivalve interactions to complex models in which multiple ecosystem processes are considered (Figure 1.1 in Cranford et al. 2006). Hydrodynamics are parameterized as integrated water exchange coefficients among large areas of the domain in box models (Dowd 2005) or by coupling a hydrodynamic model to the biogeochemical one. The hydrodynamic model can be 2D or 3D depending on the characteristics of the modeled system, and the coupling can be performed offline or online (Filgueira et al. 2012). In offline coupling techniques the physical model is run first and the biogeochemical model is run subsequently using the physical outputs from the different time steps. By contrast, online coupling dictates that both hydrodynamic and biogeochemical models are performed simultaneously. The complexity of the model in both biogeochemical and hydrodynamic submodels can be simplified depending on the objectives of the study. For example, Guyondet et al. (2013) coupled a 2D hydrodynamic model to a simplified biogeochemical model in which only seston was considered, and parameterized the balance between seston production and bivalve consumption to explore food depletion and optimize shellfish culture. Dowd (2003) limited spatial resolution to a longitudinal transect of a bay with concomitant simplification of hydrodynamics. Therefore, although model complexity varies significantly, box and fully-spatial models have been used to explore both production and ecological CC. Examples of such investigations are listed in Table 4.

Most CC models have focused on the dynamics of phytoplankton or seston and their interaction with bivalves. Specifically, most studies have examined to which extent bivalves deplete these food resources and therefore become susceptible to reduced growth (Grant and Filgueira 2011). This assessment is often carried out within the context of production CC. However, food depletion can also be used as a benchmark for ecological CC, given that phytoplankton constitutes the primary step in marine food webs and that their preservation is an important tenet of ecosystem-based management (Crowder and Norse 2008). The main shortcoming regarding application of these models is that the criterion or threshold for whether CC has been reached is typically subjective (Grant and Filgueira 2011). In the case of production CC, the definition of the threshold depends on a farmer’s judgment, which obviously relies on economic criteria as well as social values.

The definition of objective thresholds for ecological CC is still in its infancy and, in common with all questions in conservation ecology, the challenge lies in the quantitative identification of a conservation problem, that is, the precise definition of the limits at which ecosystem health is not compromised (Duarte et al. 2003; Fisher et al. 2009). These limits are known as tipping points, or the critical thresholds at which a small perturbation can qualitatively alter the state or development of a system (Lenton et al. 2008). When a perturbation is beyond a tipping point, the resilience of the system is exceeded and the system reorganizes (Crowder and Norse 2008), altering ecosystem functioning and consequently ecosystem services. Precautionary definitions of these tipping points are crucial in order to optimize ecosystem functioning. Grant and Filgueira (2011) have defined objective precautionary thresholds based on the natural variability of phytoplankton biomass, based on the premise that cultivated bivalves should not be allowed to graze primary producers down to a level outside their natural variability range. In other words, these thresholds consider whether aquaculture signals can be detected against the ecosystem background noise (Ferreira et al. 2013).

Tett et al. (2011) have defined a framework for CC and assimilative capacity based on dose-response curves following a DPSIR methodology (Drivers-Pressure-State-Impact-Response, Luiten 1999). Other ecological status evaluation methods such as ASSETS (Assessment of Estuarine Trophic Status, Bricker et al. 2003) or DDPSIR (Delta DPSIR) have also been applied to CC studies (e.g. Ferreira et al. (2007) and Nobre (2009), respectively). In Tett’s framework (Figure 1 in Tett et al. 2011), CC is defined as the stock that can be kept and the harvest that can be extracted without damaging the health of the ecosystem or interfering with other human uses of that system. This statement highlights how several components of CC can converge into a common definition. Setting thresholds is also complicated by the fact that an ecosystem’s response to a disturbance may be an increase in variability, such that no change is observed in the mean values (Cranford et al. 2012). Such issues have led to thresholds of potential concern (TPC), which are a set of operational goals along a continuum of change for selected environmental indicators. TPC values can change when new ecological information is available, allowing managers to distinguish normal ‘background’ variability from a significant change (Cranford et al. 2012).

FOOD WEB MODELS

Mass-balance food web models have also been used to explore the influence of bivalve aquaculture on food web dynamics. The most popular tool for food web modelling is EwE , which has three main components: Ecopath – a static, mass-balanced snapshot of the system; Ecosim – a time dynamic simulation module for policy exploration; and Ecospace – a spatial and temporal dynamic module primarily designed for exploring impact on and placement of protected areas. The combination of these three different components of EwE allows for solving different research questions by adjusting the approach/complexity of the model. Early models were developed exclusively in Ecopath, providing a static snapshot of the ecosystem, but with limited applicability due to the impossibility of simulating dynamic processes through time (Grant and Filgueira 2011; McKindsey 2013), which is key for coastal zones. In addition, Ecopath was not able to provide spatially-explicit outcomes, another handicap for its applicability (McKindsey et al. 2006). However, the addition of Ecosim and Ecospace avoids these limitations. EwE outcomes provide valuable information about ecosystem functioning in terms of energy flows towards different species across several trophic levels (e.g. Wolff 1994; Wolff et al. 2000; Taylor et al. 2008). Although Wolff (1994) explored CC with Ecopath, three main studies, Jian and Gibbs (2005) and Byron et al. (2011b and 2011c) have specifically focused on production and ecological CC and all of them reached the same conclusion: production CC, defined as the level of production at which the trophic web is reduced to nutrient-phytoplankton-bivalve, is higher than ecological CC, defined as the level of culture at which there are no major changes in the structure and energy fluxes of the food web. The main advantage of food web modelling is that it allows the study of many species and trophic levels at the same time, a task that is very difficult to achieve with any of the models previously described. Therefore, this modelling approach is very useful when species from different trophic levels are involved in the CC goals, such as in the fishery of an economically/socially relevant species. The drawback is that EwE uses a largely top-down mass-balance approach and poorly represents bottom-up effects, which are critical in bivalve aquaculture sites due to the impact of cultured biomass on nutrients and detritus (McKindsey 2013).

CONCLUSIONS

Impacts related to bivalve aquaculture may occur in both benthic and pelagic environments. It appears that direct impacts on the benthic environment are limited to the vicinity of the farm (Hargrave et al. 2008; Cranford et al. 2009; Guyondet et al. 2014). Impacts on the pelagic environment may trigger larger ecosystem effects, potentially affecting phytoplankton populations and concomitantly higher trophic levels that depend on phytoplankton production. CC is one of the most useful concepts for assessing the impact of bivalve aquaculture development. The different components of CC allow the inclusion of ecological, economical and societal elements into the assessment. For this reason multiple stakeholders, including scientists, farmers, regulators, NGOs, etc., must take part in an integrated management process (Byron et al. 2011a). Two CC components, production and ecological, are typically investigated using mathematical models that integrate complex interactions between aquaculture and the environment. Due to the significant influence of local environmental conditions on ecosystem functioning, CC studies are site specific (Cranford et al. 2012). Spatially-explicit models are particularly desirable due to their more accurate description of complex hydrography and straightforward applications of outcomes to marine spatial planning processes. Nevertheless, spatially-explicit models demand a high level of complexity, which in turn can increase scientific uncertainty of the outcomes. Accordingly, the modelling approach as well as spatial scale and resolution of the model must be adjusted to the goals of the study. For example, the Canadian Fisheries Act emphasizes protecting the productivity of commercial, recreational and aboriginal fisheries (DFO 2014). Thus, exploring ecological CC under the umbrella of this goal would call for (1) an analysis of energy transfer through the trophic web using food web modelling; and/or (2) a strong focus on the dynamics of phytoplankton, which constitutes the primary step in marine food webs, when using other modelling approaches. Therefore, indices, farm models, spatial models and food web models are useful tools for exploring CC and all of them present advantages and disadvantages. The main challenge in the estimation of ecological CC is the definition of acceptable/unacceptable ecological impacts. Identifying tipping points of ecological resilience is critical for identifying thresholds and advancing the application of ecological CC.

ACKNOWLEDGEMENTS

This research was funded by the Canadian Department of Fisheries and Oceans under the Program for Aquaculture Regulatory Research (Project PARR-2011-G-04). The authors gratefully acknowledge Lori Cuddy, Michael Cherry and Thomas Landry for helpful discussions.

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TABLES

Table 1. Activities related to bivalve culture that may influence the ecological carrying capacity in coastal areas (from McKindsey 2013).

Aquaculture process	Aquaculture activity	Impact on ecosystem
Seed collection	Dredging	Disturbance of benthic communities, especially the removal of long-living species Removal of juveniles from wild populations of target species Collection of non-target species Suspension of sediments Depletion of food resources for other species Release of H2S and reduction of dissolved oxygen in the water due to oxygen-consuming substances, release of nutrients
	Artificial collectors	Removal of juveniles from wild population of target species Increasing target and non-target species recruitment success Alteration of the hydrodynamic regimes Acting as fish aggregation devices Risk of entanglement for large vertebrates (e.g., marine mammals, sea birds, turtles, sharks) Foci for nuisance species
	Hatcheries	Chemical pollution (e.g., pharmaceuticals) Genetic selection Spread of diseases
	Importation	Introduction of alien species Genetic pollution Spread of diseases
Ongrowing	Effects common to all techniques	Organic enrichment of seafloor Providing reef-like structures Alteration of hydrodynamic regime (current speed, turbulence) Food web effects: competition with other filter feeders, increasing recycling speed of nutrients, removal of eggs and larvae of fish and benthic organisms Spawning: release of mussel larvae Providing food for predators of bivalves Control of predators and pests
	Bottom culture	Activities to prepare the culture plots (e.g., dredging for predator removal) Placement of protective structures (netting, pipes) Removal of associated organisms by dredging and relaying Competition for space with wild benthos organisms
	Artificial structures for suspended and off-bottom culture (trestles, poles, rafts, longlines)	Acting as artificial reef or fish aggregation devices (attraction/displacement or enhancement of animals) Risk of entanglement for large vertebrates (e.g., marine mammals, sea birds, turtles, sharks) Foci for nuisance species
Harvesting	Effects common to all techniques	Removal of biomass, nutrients Removal of filtration capacity Removal of non-target species Competition with predators
	Dredging	Disturbance of benthic communities, especially removal of long-living species Suspension of sediments Release of H2S and decrease of dissolved oxygen in the water due to oxygen-consuming substances, release of nutrients
	Collection of off-bottom structures	See above
Processing	Dumping of by-catch Relaying near auction houses Depurating	See above

Aquaculture process	Aquaculture activity	Impact on ecosystem
	Dumping of shells Effluents from processing plant Spread of alien species or diseases	

Table 2. Residence time (RT), clearance time (CT), phytoplankton turnover time (PT), CT/RT and CT/PT indices for different embayments of Prince Edward Island, Québec and Nova Scotia. This table summarizes preliminary data (Cranford et al. (unpublished manuscript)).

Region	Site	Residence Time (RT, day)	Clearance Time (CT, day)	Phytoplankton Turnover Time (PT, day)	CT/RT	CT/PT
P.E.I.	Boughton	1.87	4.07	1.70	2.18	2.39
	Brudenell / Montague	2.20	5.78	1.70	2.63	3.40
	Cascumpeque	1.59	130.10	1.48	81.9	87.9
	Cardigan	2.24	1.83	1.70	0.82	1.08
	Malpeque	2.13	16.69	1.70	7.82	9.82
	Murray	1.25	2.41	1.48	1.93	1.63
	New London	2.37	6.47	1.48	2.73	4.37
	Savage	1.52	1.63	1.48	1.07	1.10
	St. Mary’s	1.33	2.97	1.70	2.24	1.75
	St. Peter’s	3.08	2.45	1.48	0.80	1.65
	Tracadie	2.19	1.84	1.48	0.84	1.24
Québec	Grand Entrée	2.17	6.94	0.73	3.20	9.51
Nova Scotia	Lunenburg	2.43	430.12	0.79	177.4	544.5
	Ship	3.21	23.92	0.79	7.46	30.28
	St. Ann’s	5.70	33.25	1.79	5.83	18.58
	Whitehead	4.14	196.29	0.79	47.4	248.5

Table 3. Set of desirable properties when assembling a modelling system to address aquaculture carrying capacity (from Ferreira et al. 2013).

Property	Description
1	No single model solves all problems. Overparameterization and code bloat only make matters worse.
2	Models should only be as complex as the problem requires. In other words, as simple as possible. Increased computational power is no excuse for unnecessary complexity.
3	Models should be able to work independently, present value in doing so, and add further value when working in conjunction with other models.
4	Models should define exactly what problems they can address, as part of the overall questions for an ecosystem, rather than the opposite.
5	Any model in the system must be able to receive input from data or from other models and be able to supply outputs in a form that can be easily used by other models.
6	Different models are appropriate for different scales in space and time. Carrying capacity assessment may require scales as short as a tidal cycle (e.g., for intertidal culture of clams) and as long as a decade (e.g., for coupling ecological models with economic models).
7	Models share a challenge with field sampling with respect to the conversion of data (measured or modeled) into information that is useful for managers; the use of screening models, or other approaches that help to distil data into meaningful information, is a vital component of any system.

Table 4. Publications that present Box models and fully-spatial models for assessing production and ecological carrying capacity (CC).

<table>
<tr><th>Type of model</th><th>Production CC[1]</th><th>Ecological CC[2]</th></tr>
<tr><td rowspan="2">Box model</td><td>Raillard and Ménesguen 1994
Dowd 1997
Bacher et al. 1998
Ferreira et al. 1998
Duarte et al. 2003
Grant et al. 2007
Filgueira et al. 2010</td><td>Chapelle et al. 2000
Dowd 2005
Filgueira and Grant 2009</td></tr>
<tr><td colspan="2">Nunes et al. 2011</td></tr>
<tr><td rowspan="2">Fully-spatial model</td><td>Duarte et al. 2003
Spillman et al. 2008
Maar et al. 2009
Ibarra et al. 2014</td><td>Cugier et al. 2008
Brigolin et al. 2009
Grangeré et al. 2010
Filgueira et al. 2013
Filgueira et al. 2014b</td></tr>
<tr><td colspan="2">Grant et al. 2007
Ferreira et al. 2008
Guyondet et al. 2010
Dabrowski et al. 2013
Filgueira et al. 2014a
Guyondet et al. 2014</td></tr>
</table>

[1] Some models do not focus specifically on production CC but on ecosystem effects or culture-environment interactions, which ultimately would allow the exploration of ecological CC.
[2] Some models do not focus specifically on ecological CC but on aquaculture practices and their effects on production.

Aquaculture 298 (2009) 1–15

Contents lists available at ScienceDirect

Aquaculture

journal homepage: www.elsevier.com/locate/aqua-online

Bivalve aquaculture in estuaries: Review and synthesis of oyster cultivation effects

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ARTICLE INFO

Article history:
Received 11 March 2009
Received in revised form 28 September 2009
Accepted 29 September 2009

Keywords:
Aquaculture impact
Pacific oyster
Crassostrea gigas
Suspended culture
Risk assessment

ABSTRACT

Oyster farming in estuaries is a globally important industry based primarily around the Pacific oyster *Crassostrea gigas*, for which a common technique is elevated culture on racks, trestles and other structures. We review literature on cultivation impacts, revealing a research focus and state of knowledge that largely parallels that for other aquaculture species and cultivation methods. Ecological studies of elevated culture effects have focused on changes to the benthos from biodeposition, and largely show that impacts are localized and minor by comparison with many other forms of aquaculture. The broader ecological issues associated with elevated oyster culture include the effects of pests (fouling pests, toxic/noxious microalgae, disease), creation of novel habitat (e.g. by fouling of farm structures and accumulation of shell), alteration to nutrient cycling, depletion of suspended particulate matter by oyster crops, and related effects on higher trophic level animals including fish, seabirds and marine mammals. These issues are less well understood for elevated culture systems, but ecological effects can be inferred from the few studies that have been conducted, from other forms of bivalve aquaculture (e.g. mussels), and to some extent from fundamental knowledge of the role of oysters as 'ecosystem engineers'. We use a risk ranking method to evaluate ecological risks (and associated uncertainty intervals) for each of the issues associated with estuarine oyster culture, based on subjective assessment of the likelihood and consequences (severity, spatial extent and duration) of adverse effects. Our assessment reveals that the introduction and spread of pest species are potentially important but often overlooked consequences of oyster cultivation. By comparison with most other sources of impact, the spread of pests by aquaculture activities can occur at regional scales, potentially leading to ecologically significant and irreversible changes to coastal ecosystems. We suggest that future studies of cultivation effects redress the balance of effort by focusing more on these significant issues and less on the effects of biodeposition in isolation. Furthermore, the acceptability of aquaculture operations or new developments should recognize the full range of effects, since adverse impacts may be compensated to some extent by the nominally 'positive' effects of cultivation (e.g. habitat creation), or may be reduced by appropriate planning and management. Even more broadly, aquaculture developments should be considered in relation to other sources of environmental risk and cumulative impacts to estuarine systems at bay-wide or regional scales, so that the effects of cultivation are placed in context.

Contents

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doi:10.1016/j.aquaculture.2009.09.032

1. Introduction

Intertidal oyster cultivation is one of the most important aquaculture industries globally (FAO, 2006a). While this industry sector is based on a range of species, Pacific oysters (*Crassostrea gigas*) are by far the most dominant (>96% by value and tonnage; FAO, 2006a,b,c), having been spread either deliberately or inadvertently (e.g. via shipping) to many countries (Kaiser et al., 1998; Dumbauld et al., 2009). Oyster cultivation takes place primarily on the tidal flats of estuaries, using farming methods that differ among localities according to environmental conditions, the type of product marketed, and tradition (FAO, 2006a). A common technique is elevated (off-ground) culture, which typically involves laying oysters on sticks, in mesh bags or trays across wooden racks or steel trestles (~0.3–1 m high) that are fixed in the intertidal zone and exposed during low tide, or uses stakes or long-lines (e.g. Forrest and Creese, 2006; Dubois et al., 2007; Leguerrier et al., 2004; McKindsey et al., 2006). Depending on region and the reliability of natural settlement, seed-stock may be derived from seabed populations, from wild-caught spat on artificial collectors, and increasingly from hatchery brood-stock (McKindsey et al., 2006; Dumbauld et al., 2009).

The occupation of space by intertidal structures means that oyster cultivation can conflict with a range of other environmental, social and economic values (DeFur and Rader, 1995; Simenstad and Fresh, 1995; Kaiser et al., 1998; Read and Fernandes, 2003). The literature on environmental effects is dominated by papers that describe changes to sediments and associated infaunal assemblages beneath cultivation areas (e.g. Ito and Imai, 1955; Kusuki, 1981; Mariojouls and Sornin, 1986; Castel et al., 1989; Nugues et al., 1996; Spencer et al., 1997; De Grave et al., 1998; Kaiser et al., 1998; Forrest and Creese, 2006; Dubois et al., 2007; Bouchet and Sauriau, 2008). In addition to benthic effects, there are a range of broader ecological issues associated with elevated oyster aquaculture that are less well-recognized or need to be considered in a comparative context (Fig. 1). These include the introduction of pests (fouling pests, toxic/noxious microalgae, disease), creation of novel habitat, alteration to water flows and nutrient cycles, and depletion of suspended particulate matter (especially phytoplankton) by oyster crops (ICES, 2005a; McKindsey et al., 2006). Related considerations are the wider ecosystem consequences of such changes, for example implications for fish, seabirds and marine mammals.

With some exceptions, knowledge of this broad range of ecological effects from oyster aquaculture is limited. Furthermore, where the ecological effects of elevated culture methods are specifically addressed, the complexity of some of the ecosystem issues and interactions depicted in Fig. 1 means they are often described in the literature only superficially (e.g. Crawford, 2003). Alternatively more thorough assessments have usually focused on a subset of the broad range of potential issues (e.g. McKindsey et al., 2006; Dumbauld et al., 2009). We propose that to understand and manage ecological risks from elevated culture systems, there is a need for a more integrated and in-depth assessment in which the relative significance of each issue is considered within the context of the full range of actual or potential ecological effects.

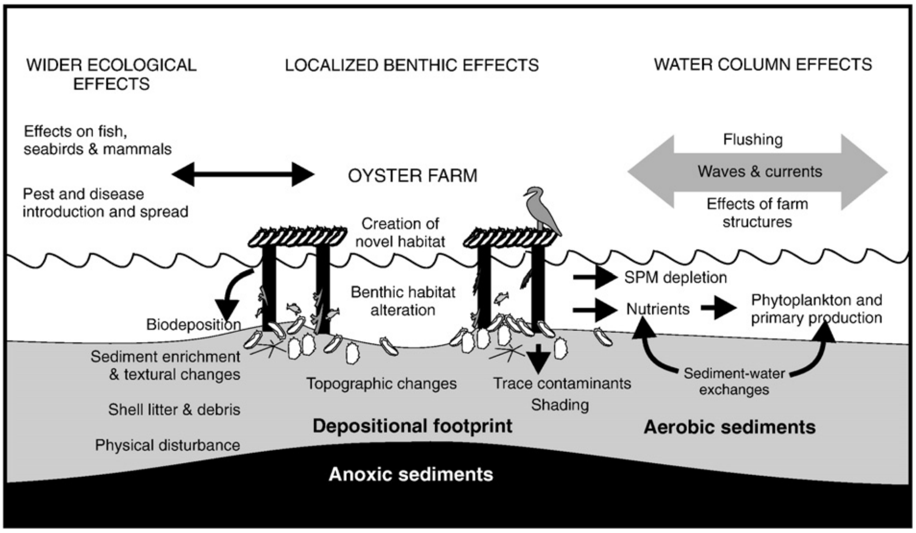

Fig. 1. Schematic of actual and potential ecological effects from elevated intertidal oyster cultivation. SPM=suspended particulate matter.

As a contribution to such a goal, this paper provides a synthesis of the ecological impacts that can arise with the development of elevated oyster cultivation in estuaries, and assesses the relative importance of the different effects. To provide insight into effects for which little is known, we expand our synthesis to include other types of aquaculture (especially of other bivalves) for which the suite of ecological issues is qualitatively similar. We also draw on the literature on aquaculture effects more broadly (i.e. for different culture methods or species) where this assists in placing in context the magnitude of effects from oyster cultivation. Similarly, where valid comparisons can be made, we refer to the substantial body of knowledge for natural or restored oyster reefs that describes the functional role of oysters as 'ecosystem engineers'; such comparisons can facilitate understanding of the potential for wider ecosystem changes from cultivation. We then discuss the relative ecological importance of the different ecological issues, and highlight key knowledge gaps or uncertainties. Based on these findings, research and management implications are then discussed. While we recognize that a range of short-term ecological effects may arise as a result of oyster farm construction, and in relation to other aspects of farming operations such as off-site spat catching and product processing (McKindsey et al., 2006), we restrict discussion in this paper to the sea grow-out stage of oyster aquaculture.

2. Local benthic effects

2.1. Biodeposition and enrichment

Oyster farms act as biological filters that concentrate suspended particulate matter from the water column as it flows through the culture, producing waste particles in the form of faeces and pseudofaeces. These wastes (generally referred to as 'biodeposits') are heavier than their constituent particles, and readily settle on the seabed beneath culture areas (Haven and Morales-Alamo, 1966; Kusuki, 1981; Mitchell, 2006). Since biodeposits are organic-rich and consist of a substantial proportion of fine particles (i.e. silt and clay), seabed sediments beneath oyster cultures can become organically enriched and fine-textured relative to surrounding areas, and have anoxic sediments closer to the sediment surface (Forrest and Creese, 2006).

Changes in physico-chemical characteristics beneath oyster cultures can lead to a displacement of large-bodied macrofauna (e.g. heart urchins, brittle stars, large bivalves) and the proliferation of small-bodied disturbance-tolerant 'opportunistic' species (e.g. capitellid polychaetes and other marine worms). In some instances an associated reduction in the richness of the infaunal assemblage has been described for elevated cultures (e.g. Castel et al., 1989; Nugues et al., 1996), consistent with moderate organic enrichment in terms of the classic conceptual model of Pearson and Rosenberg (1978). However, in many case studies the response of the infaunal assemblage has been less pronounced, for example evident as a change in species composition and dominance without an appreciable effect on richness (Forrest and Creese, 2006). Yet other studies have revealed little or no discernible enrichment effect on infauna (e.g. Crawford et al., 2003). In fact, extreme enrichment effects as a result of oyster farming have been described historically only for suspended culture systems in Japan, and been attributed to repeated culturing and overstocking (Ito and Imai, 1955; Kusuki, 1981). Hence, it is apparent that the magnitude of benthic enrichment from elevated intertidal culture is generally relatively minor by comparison with suspended subtidal culture of fish (e.g. Brown et al., 1987; Karakassis et al., 2000; Forrest et al., 2007a) and, to a lesser extent, other bivalves (e.g. Mattsson and Lindén, 1983; Kaspar et al., 1985; Grant et al., 1998). Irrespective of the magnitude of the effect, without exception it is apparent that direct benthic effects associated with oyster cultivation are highly localized to farmed areas (extending tens of metres or less from structures) and can be greater directly beneath cultivation structures than in the space between them (Forrest and Creese, 2006).

The magnitude of effects from enrichment will depend primarily on stocking density and biomass in relation to the flushing characteristics of the environment (Pearson and Black, 2001). Additionally, the level of biodeposition for a given stocking density, and the assimilative capacity of the environment, may vary seasonally (Kusuki, 1981; Souchu et al., 2001; Mitchell, 2006). To our knowledge, however, the relative role of these different attributes has not been quantified for oyster farms. In the case of intertidal culture, the capacity of the environment to assimilate and disperse farm wastes will mainly depend on water current velocity and wave action (Souchu et al., 2001), as these factors control the size and concentration of the depositional 'footprint'. Increased flushing from currents and waves will reduce biodeposit accumulation and increase oxygen delivery to the sediments, thus allowing for greater assimilation of farm wastes (Findlay and Watling, 1997; Mitchell, 2006). Negligible enrichment from elevated oyster farms in Tasmania has been attributed to a combination of low stocking densities and adequate flushing (Crawford, 2003; Crawford et al., 2003; Mitchell, 2006). Generally, well-flushed aquaculture sites can be expected to have depositional footprints that are less intense but more widely dispersed than shallow or poorly flushed sites (Pearson and Black, 2001).

Recovery rates of seabed communities from oyster farm biodeposition and enrichment are not well understood, but in some localities (e.g. where coarse sandy sediments are prevalent) appears to be relatively rapid (time periods of months) once farming ceases (Martin et al., 1991). Rapid recovery can also be inferred from seasonal or temporal reductions in the severity of benthic enrichment effects over time scales of months in unvegetated soft-sediment habitats (Forrest, 1991). By comparison, time scales of recovery in strongly enriched (sometimes near-azoic) muddy sediments beneath fish farms can be highly variable, but may be many years at poorly flushed subtidal sites (Karakassis et al., 1999; Pereira et al., 2004; Forrest et al., 2007a).

2.2. Accumulation of shell litter, debris and associated organisms

The accumulation of live oysters, shell litter and farm debris (e.g. oyster growing sticks), and fouling or epibenthic organisms beneath grow-out structures can be a highly visible effects of oyster farms during low tide (Fig. 2). The extent of drop-off to the seabed is likely to depend on the type of cultivation system (e.g. stick culture is likely to deposit more debris than basket or cage culture) and may be exacerbated periodically during harvesting. The degree of fouling accumulation will depend on the degree to which structures become fouled, and patterns of natural drop-off or active defouling by farm personnel. Subsequent effects to benthic community composition, for example aggregation of carnivorous and deposit feeding species in response to the food supply (e.g. sea stars) and competition between deposited shellfish and benthic filter-feeders, are indicated for other forms of bivalve aquaculture (Inglis and Gust, 2003; Smith and Shackley, 2004; Amours et al., 2008; Hartstein and Rowden, 2008) and conceivably occur in the case of elevated oyster culture. Excessive deposition and decay of fouling biomass may also exacerbate the organic enrichment described above, although such effects would likely be patchy beneath cultivation areas and were not evident in a recent study at a well-flushed location in eastern Canada (Mallet et al., 2009).

Hard surfaces on the seabed such as live and dead oysters, calcareous debris (e.g. bivalve shells, serpulid polychaete tubes) and farm materials potentially provide novel habitats for fouling organisms and associated mobile biota, which would otherwise not occur (or be at reduced densities) in the absence of oyster growing. Such effects have been widely documented in the case of on-ground shellfish culture (Dumbauld et al., 2001; Hosack et al., 2006; Powers

Fig. 2. Fouling by oysters and other organisms on derelict oyster racks. Altered topography and rows of accumulated shell and debris are visible in the background (photo courtesy of B. Howse, Northland Regional Council, New Zealand).

et al., 2007; Ysebaert et al., 2009) and oyster reefs (Peterson et al., 2003; Escapa et al., 2004; Ruesink et al., 2005; Coen et al., 2007). The structured habitat provided by oyster reefs can support a diversity of taxa (macroalgae, sessile and mobile invertebrate epifauna, infauna, fish, birds) that may be absent or at reduced densities in adjacent unvegetated soft-sediment habitats (Ruesink et al., 2005 and references therein). Similarly, oyster shell has been used to successfully enhance estuarine habitat for juvenile Dungeness crabs (*Cancer magister*), to compensate for habitat loss caused by dredging (Dumbauld et al., 2000). Probably the main factors that would limit the value of hard substrata deposited beneath operational oyster farms would be the effect of enhanced sedimentation beneath structures, or sediment resuspension and physical disturbance from farming activities (see below). Nonetheless, the introduction of novel habitats could result in fundamental or long-term shifts in benthic community composition after cessation of farming, depending on site-specific variation in environmental conditions, oyster species and density, and the extent and persistence of accumulated material. Dumbauld et al. (2000) observed that transplanted oyster shell could sink or become covered in sediment in a matter of months. However, in long-established cultivation areas the longer term persistence of shell and other inorganic material is often evident (Fig. 2), although regulatory authorities in many countries are increasingly stipulating management practices to mitigate such effects (e.g. requiring removal and land disposal of accumulated material).

2.3. Changes in seabed topography and sedimentation

Changes in seabed topography (in the order of a few tens of centimetres at maximum) have been described beneath oyster farms in several studies (Ottmann and Sornin, 1982; Everett et al., 1995; Forrest and Creese, 2006; see also Fig. 2). Such changes can result from the accumulation of shell and inorganic debris, and erosion or accretion of sediment beneath and between farm structures (Forrest and Creese, 2006). Sedimentation rates directly beneath cultures are generally elevated by comparison with non-culture areas (Mariojouls and Sornin, 1986; Sornin et al., 1987; Nugues et al., 1996), being as much as three times greater directly beneath farm structures than at control sites (Forrest and Creese, 2006). However, effects on seabed topography are more likely to be related to changes in hydrodynamic conditions caused by the structures themselves rather than enhanced sedimentation (Kirby, 1994). Excessive sediment build-up within Pacific oyster leases can occur at sites where cultivation structures are in high density or aligned perpendicular to tidal currents, resulting in the entrapment of suspended sediments (Kirby, 1994; Handley and Bergquist, 1997). In such instances oyster leases in New Zealand have become un-useable and farming abandoned, with shell litter and debris still evident many years later. Redistribution of sediments either into (Kirby, 1994) or out of (Mallet et al., 2009) culture cites may also occur in relation to events such as storms that lead to large scale sediment mobilisation.

2.4. Physical disturbance

Physical disturbance is obviously an issue with on-ground shellfish culture and harvest methods (Dumbauld et al., 2009). At least two studies of elevated oyster culture have also recognized physical disturbance, in particular from vessel movements (e.g. propeller wash) and farm personnel walking between cultivation structures, as having a strong influence on benthic changes beneath farm sites (De Grave et al., 1998; Forrest and Creese, 2006). Forrest and Creese (2006), for example, described a relatively strong association between benthic macrofaunal composition and decreased sediment shear strength beneath Pacific oyster cultures in New Zealand, which they suggested could reflect physical disturbance beneath racks. Impacts from physical disturbance are conceivably equally as important as enrichment within elevated cultivation areas, and perhaps more important where enrichment is negligible. Despite this, the relative importance of these two effects is yet to be rigorously evaluated in the case of elevated culture; hence the recovery rate of seabed communities from disturbance effects is unknown. Studies of on-ground culture systems have more clearly demonstrated physical effects during intermittent shellfish harvesting, and the recovery of soft-sediment communities in a matter of weeks to months in unvegetated habitats (McKindsey et al., 2006 and references therein). By contrast, recovery from physical disturbance by eelgrass (*Zostera marina*) may take several years (McKindsey et al., 2006; Dumbauld et al., 2009 and references therein).

2.5. Shading

Shading by farm structures could reduce the amount of light reaching the seafloor, with implications for the growth, productivity, survival and depth distribution of ecologically important primary producers such as benthic microalgae, macroalgae or seagrasses. In the context of studies that report negligible effects on seagrass beneath oyster farms (Crawford, 2003; Ward et al., 2003), we can infer that shading effects in such cases are of little significance. However, other studies have described adverse effects on seagrass beneath oyster racks and suggested shading as a possible cause (e.g. Everett et al., 1995). To our knowledge, however, the relative importance of shading versus other sources of seabed impact has never been conclusively established, and to do so would require targeted manipulative experiments. Despite the absence of clear evidence for adverse effects from shading, Hewitt et al. (2006) demonstrated that a small reduction in cover of New Zealand eelgrass (*Zostera muelleri*) was theoretically possible because of shading from planned long-line oyster cultures. Shading effects are conceivably important where oyster farms are placed across seagrass and algal habitats in environments of relatively high water clarity, and in locations (e.g. well-flushed systems) where other ecological effects (especially those from sedimentation and biodeposition) are minimal. Alternatively, the incremental reduction in incident light by shading may be more important in turbid systems where the depth distribution of benthic algae and macrophytes is already light limited. Clearly the potential for adverse effects is situation-specific, but can to a large extent be mitigated by appropriate site selection and management (Dumbauld et al., 2009).

2.6. Contaminant inputs

Operational oyster farms do not generally require the ongoing input of external materials that could introduce trace contaminants to the marine environment, as can occur for example as a result of synthetic feed inputs to sea-cage fish farms (Morrisey et al., 2000; Easton et al., 2002; Schendel et al., 2004). Possible exceptions to this situation arise in shellfish aquaculture where compounds such as hypochlorite and acetic acid have been used to mitigate the effects of biofouling (Carver et al., 2003; Forrest et al., 2007b); however, these tend to be non-persistent contaminants whose use is unlikely to lead to significant non-target effects (Locke et al., 2009). Historically, many oyster cultivation racks have been constructed from wood treated with preservatives (e.g. copper–chromium–arsenic, CCA; creosote) that could leach into surrounding waters. Highly localized effects on sediments have been described in the vicinity of marine pilings as a result of CCA leaching (Weis et al., 1993), consistent with expectations that trace metals that are released to the water column will rapidly bind to suspended sediments and organic material (Forstner, 2005). Such binding is likely to reduce the bioavailability and toxicity to associated biota, and the release of contaminants from treated timber in seawater is reported to decrease over time (Brooks, 1996; Breslin and Adler-Ivanbrook, 1998). Hence, this issue is probably of negligible significance in the case of oyster culture sites where wooden structures are used. We also note that there is an increasing trend to use alternative construction materials or develop strict regulatory guidelines around the use of treated timber (e.g. DPI, 2008).

3. Water column effects and interactions with the benthic environment

3.1. Altered currents and flushing

Currents and waves play an important role in ecosystem function. In relation to shellfish farming these include delivery of seston and dissolved oxygen, and the flushing of wastes and associated nutrients into and out of the localized environment. For example, excessive enrichment effects on benthos could occur if currents are not above a critical threshold to allow dispersion and resuspension of seabed sediments and shellfish farm biodeposits. Although there appears to be little published information, oyster farm structures and farm-related alterations to seabed topography (e.g. from shell accumulation) are likely to lead to effects on waves, currents and flushing characteristics in the vicinity of farm sites (Gouleau et al., 1982; Nugues et al., 1996; Hewitt et al., 2006). Literature for oyster reef habitats indicates that flow changes across the seabed may alter fluxes of materials (e.g. sediments) to adjacent habitats, and influence ecological processes such as patterns of dispersal and recruitment of invertebrates and fish (Breitburg et al., 1995; Ruesink et al., 2005). Effects of this general nature are also conceivable in the case of elevated oyster culture, although specific differences can be expected given that the extent to which flows are modified will differ for different types of habitat (e.g. because of differences in the 'porosity' of benthic reef versus elevated structures), and depend on attributes of the cultivation structures (e.g. height, density) and the extent to which cultivation physically alters the seabed (e.g. by shell accumulation).

3.2. Water quality

Natural oyster reefs are considered to have the potential to improve estuarine water quality by filtering sediments and other suspended particulate matter (SPM) from the water column (e.g. Gottlieb and Schweighofer, 1996; Ruesink et al., 2005; Grizzle et al., 2006). As a consequence, there is much interest in the restoration of degraded oyster reefs as a means of top-down control of phytoplankton densities in eutrophic estuaries (Newell, 2004; Cerco and Noel, 2007; Newell et al., 2002, 2007). A recent study also suggests that elevated oyster cultures can have a comparable function (Lin et al., 2009; see below). The adverse effects of intertidal culture systems on water quality in estuarine environments are less well understood, but are likely to be relatively minor given that seabed enrichment is low and external contaminant inputs are minimal, as described above. The only published cases of adverse water quality from oyster aquaculture have arisen in suspended cultivation systems where farms have been over-stocked or located in poorly flushed environments. Early studies of suspended culture of Pacific oysters in Japan revealed adverse water column impacts that were related to excessive biodeposition (Ito and Imai, 1955; Kusuki, 1981). For example, Ito and Imai (1955) described seabed enrichment so severe that oyster culture areas became 'self-polluting' (i.e. leading to oyster mortality) as a result of dissolved oxygen depletion in the overlying water column and the associated release (from sediments) of hydrogen sulphide at toxic concentrations.

By contrast, a study in Marennes-Oléron Bay (a major Pacific oyster culture area in France) suggests that mortality can occur as a result of a range of factors, and not simply a negative feedback on water quality (Soletchnik et al., 2005). The findings of the latter study further indicate that the potential for adverse water quality-related effects in the case of elevated intertidal culture is low, which is perhaps not surprising considering that intertidal farm sites are substantially or completely flushed on every tidal cycle. Any water quality effects associated with elevated culture can undoubtedly be minimized by appropriate site selection and farm design (e.g. ensuring that farm structures are configured in a way that causes minimal retardation of flushing processes).

3.3. Nutrient cycling

The effect of elevated oyster cultivation on nutrient cycling in estuaries is incompletely understood, highly complex and situation-specific. Based on information from other bivalve culture systems, and natural or restored oyster reefs, it is evident that effects will be determined by processes involving filter-feeding and dissolved nutrient excretion, biodeposition and sediment remineralization of nutrients, and loss of nutrients through oyster harvest (Prins et al., 1998; Newell, 2004; Porter et al., 2004; Su et al., 2004). The production of dissolved (hence bioavailable) nutrients can occur directly via excretion by the oyster stock, or indirectly via remineralization and subsequent release from enriched sediments (Souchu et al., 2001; Dumbauld et al., 2009). The resultant effects of dissolved nutrient release on algal production involve complex interactions that are likely to be highly variable in relation to factors such as flushing, temperature, water clarity, stocking density, and the level of seabed enrichment. For example, although oysters may deplete phytoplankton, dissolved nutrients released from oyster excretion or sediment remineralization have the potential to offset this effect by simultaneously stimulating phytoplankton production at local scales (e.g. Prins et al., 1998; Pietros and Rice, 2003). Conversely, in the case of oyster reefs where filter-feeding leads to locally increased water clarity (Cerco and Noel, 2007), the production of benthic algae and seagrasses may be enhanced, thereby reducing the flux of dissolved nutrients to the water column and reducing phytoplankton production (Souchu et al., 2001; Newell, 2004; Porter et al., 2004). For example, increased water clarity resulting from restoration of oyster reefs is predicted to lead to an increased biomass of submerged aquatic vegetation (Newell and Koch, 2004; Cerco and Noel, 2007). For elevated culture systems, however, decreased sediment shear stress beneath grow-out areas (i.e. indicative of greater erosion potential), combined with turbulence induced by culture structures, may lead to enhanced sediment resuspension and high turbidity (Forrest and Creese, 2006; Leguerrier et al., 2004).

3.4. Suspended particulate matter depletion and ecological carrying capacity

Oysters can filter particles within the 4–100 µm size range (Hawkins et al., 1998; Dupuy et al., 2000), and hence can derive nutrition from suspended particulate matter (SPM) in the form of phytoplankton (predominantly), detritus, bacteria, protozoa, zooplankton, and resuspended benthic microalgae (Le Gall et al., 1997; Dame and Prins, 1998; Leguerrier et al., 2004). Cultured oysters may also contribute to the SPM pool during spawning. There has been considerable research into food depletion and modelling of carrying capacity for oyster culture (e.g. Ball et al., 1997; Bacher et al., 1998; Ferreira et al., 1998) as well as for other bivalves and polyculture systems (e.g. Carver and Mallet, 1990; Prins et al., 1998; Smaal et al., 1998; Gibbs et al., 2002; Nunes et al., 2003). Typically, this work has focused on phytoplankton depletion and maximum production capacity within growing regions. In this respect a number of indicators of carrying capacity have been used, in particular water residence time in relation to bivalve clearance and primary production time within a system (e.g. Dame and Prins, 1998; Gibbs, 2007). The literature in this field primarily addresses the role of natural or cultivated bivalve populations, whereas the filter-feeding activities of fouling organisms and other biota associated with shellfish cultures can also be functionally important (e.g. Mazouni et al., 2001; Mazouni, 2004; Decottignies et al., 2007).

Influences from oyster aquaculture on estuarine carrying capacity are inextricably linked to the issues of nutrient cycling, SPM depletion, and coupling between the seabed and water column. Interactions between shellfish cultivation and the water column and seabed environments are complex (see review by Dumbauld et al., 2009). However, there is compelling evidence that bivalve aquaculture can affect nutrient cycling and the quantity and quality of SPM across a range of spatial scales (Prins et al., 1998; Cerco and Noel, 2007; Coen et al., 2007; Lin et al., 2009). Empirically, phytoplankton depletion is certainly evident at local scales in the vicinity of oyster cultures (Dumbauld et al., 2009) or intensive culture zones (Lin et al., 2009), and serial depletion among multiple adjacent farms at larger spatial scales has been described for other types of suspended bivalve culture (Gibbs, 2007; Grant et al., 2007). Top-down control of phytoplankton has similarly been described or inferred for estuarine systems where indigenous oyster populations have declined (Newell, 2004) or non-indigenous bivalves have reached high densities (e.g. San Francisco Bay; Nichols, 1985).

There is some evidence that SPM depletion by cultivation can negatively affect oyster production. For example, control of Pacific oyster growth by phytoplankton availability has been described for subtidal floating culture systems in environments with long residence times such as Thau Lagoon in southern France (Souchu et al., 2001). In relation to elevated intertidal culture, Marennes-Oléron Bay has been described as "…one of the few systems where bivalve filter-feeders have on two occasions been over-stocked and overexploited" (Dame and Prins, 1998). Marennes-Oléron Bay is a highly turbid system where bivalve clearance times are shorter than primary production and water residence times, and where resuspended benthic microalgae are an important food source (Dame and Prins, 1998). There are anecdotal reports that Pacific oyster production in New Zealand estuaries has also been limited by carrying capacity, although this has not been definitively proven (Handley and Jeffs, 2002).

The potential for wider effects on ecological carrying capacity as a result of SPM depletion by shellfish cultures is invariably situation-specific and scale-dependent in terms of the size of the cultivation area to the system in question (Anderson et al., 2006). Carrying capacity is also expected to be temporally variable, as the amount of phytoplankton and other SPM in estuaries is likely to be influenced by factors operating from tidal time scales to longer term climatic events such as El Niño Southern Oscillation cycles (Dame and Prins, 1998; Prins et al., 1998; Zeldis et al., 2000). Evidence that SPM depletion from elevated oyster culture has the potential to reach or exceed carrying capacity at bay-wide scales suggests that wider ecosystem effects are certainly possible. Such effects could conceivably arise not only as a function of direction depletion but also through alteration in SPM size spectra and phytoplankton species composition; thus the type and quality of food available to zooplankton and other consumers (Prins et al., 1998; Dupuy et al., 2000; Pietros and Rice, 2003; Leguerrier et al., 2004), with consequences for local populations of higher trophic level organisms such as fish. Food-web modelling for Marennes-Oléron Bay predicted a shift from pelagic to benthic consumers as a result of intertidal trestle cultivation of oysters, reflecting SPM depletion in the water column and enrichment of benthic meiofauna (Leguerrier et al., 2004). This prediction is supported by Lin et al. (2009), whose ECOPATH model simulations and field sampling both revealed a substantial increase in phytoplankton and zooplankton biomass and decrease in benthic infaunal biomass following the complete removal of oyster racks from a coastal lagoon where intensive oyster cultivation (up to 2932 racks km^{-2}) had been conducted previously.

4. Wider ecological effects

4.1. Habitat creation by farm structures

Marine farm structures and artificial structures in general, provide a three-dimensional reef habitat for colonisation by fouling organisms and associated biota (Costa-Pierce and Bridger, 2002). In a manner similar to that described above for the accumulation of oysters and debris, elevated shellfish aquaculture structures provide a novel habitat that can support a considerably greater biomass, richness and density of organisms than adjacent natural habitats (e.g. *Crassostrea virginica* cages, Dealteris et al., 2004; *Mytilus edulis* ropes, Murray et al., 2007; see also Fig. 2). It is also well-recognized that the biota fouling artificial structures can be quite different to that in adjacent rocky areas (Glasby, 1999; Connell, 2000), and can comprise a diverse assemblage of macroalgae and filter-feeding invertebrates (Hughes et al., 2005). Hence, there is considerable interest in the role played by artificial structures within the ecosystem, such as increasing local biodiversity, enhancing coastal productivity, and compensating for habitat loss from human activities (Ambrose, 1994; Costa-Pierce and Bridger, 2002; Hughes et al., 2005).

Many of the ecological roles of the habitat created by shellfish farming are well-recognized for on-ground oyster culture, as noted earlier in this paper. There is also evidence of a comparable role for suspended subtidal oyster culture structures (e.g. Lin et al., 2007), intertidal trestles (e.g. Hilgerloh et al., 2001) or other intertidal structures used for oyster cultivation (O'Beirn et al., 2004). From McKindsey et al. (2006), it is evident that the habitat complexity created by intertidal shellfish culture is likely to provide a range of ecosystem services including an enhanced food supply, a refuge from predation, a settlement surface, and protection from physical (e.g. water movement) and physiological (e.g. dessication) stress. Dealteris et al. (2004) conclude that oyster cages used for the grow-out stage of *C. virginica* have a habitat value that is considerably greater than non-vegetated seabed and at least equal to seagrass. It is also evident that some elevated culture systems provide a habitat that can be extensively colonised by wild or naturalized oysters, as described for *C. gigas* in western France (Cognie et al., 2006). As we further discuss below, the creation of novel habitat and the presence of cultivation structures can affect the wider ecosystem in a number of ways.

4.2. Effects on fish

The aggregation of various fish species around suspended aquaculture operations and other artificial structures is well-recognized

(Relini et al., 2000; Gibbs, 2004; Einbinder et al., 2006; Morrisey et al., 2006), reflecting the role of such structures in offering shelter from predation, habitat complexity and a food source. There has also been discussion of the potential for direct negative effects of cultured oysters and mussels on fish populations, primarily due to the consumption of fish eggs and larvae (Gibbs, 2004; McKindsey et al., 2006; Keeley et al., 2009). The association of fish with on-ground oyster culture has been described in a number of studies (see references in Grabowski, 2004 and Dumbauld et al., 2009). Similarly, in the case of on-ground clam culture in the United States, Powers et al. (2007) found that the emergent habitat provided by fouling of mesh bags led to densities of mobile invertebrates and juvenile fish that were elevated by comparison with adjacent sand flats, and comparable to natural seagrass.

Conceivably, therefore, the ecological role of elevated oyster farm structures, combined with habitat alterations from the deposition of oysters and associated debris, may affect fish populations in a number of ways. However, a body of published information from primary literature comparable to that describing the effects of oyster reef or on-ground culture systems is unavailable for elevated culture systems, and the limited information available describes changes that are often viewed as neutral or positive (rather than adverse) effects (McKindsey et al., 2006). For example, Dealteris et al. (2004) describe a greater association with submerged aquaculture gear by some fish species but not others. Similarly, Dumbauld et al. (2009) cite a thesis that revealed no overall increase in fish richness or abundance adjacent to oyster racks, but a greater prevalence of structure-oriented species. Trophic modelling in Marennes-Oléron Bay represents one of few attempts to understand the wider ecosystem role of elevated intertidal oyster (*C. gigas*) culture (Leguerrier et al., 2004). These authors suggested that oyster cultivation could increase the food supply to fish, which was predicted to occur as a result of increased meiofaunal production. Similarly, increased turbidity (e.g. induced by erosion around oyster farm structures) may provide a refuge from predation for small or juvenile life-stages of fish (Chesney et al., 2000; Leguerrier et al., 2004). A field mesocosm study of Pacific oyster cultivation effects in western France showed that the microhabitat created beneath trestles was more frequented by flatfish than adjacent homogenous habitat (Laffargue et al., 2006). More recently, an experimental scale deployment of oyster cages suggested that aquaculture gear could benefit populations of ecologically and economically important fish and epibenthic macrofauna in a way comparable to oyster reef habitat (Erbland and Ozbay, 2008). Similarly, the Lin et al. (2009) study of the system-wide effects of oyster rack removal described an unexpectedly large decline in the biomasses of zooplanktivorous and piscivorous reef fish post-removal. These authors suggested that the oyster racks might have previously attracted and benefited reef fish by reducing predation or enhancing their food sources.

4.3. Effects on seabirds

Effects on seabirds from elevated oyster culture conceivably arise due to the alteration of food sources, displacement of foraging habitat, and as a result of disturbance (e.g. noise) related to farm activities (Kaiser et al., 1998; Connolly and Colwell, 2005). The additional issue of entanglement has been widely discussed in relation to other forms of aquaculture or fishing practice (Butler, 2003; Bull, 2007), but is unlikely to be an important consideration for intertidal oyster culture where primarily rigid structures are used. Similarly, the effects of plastic and other marine debris on seabirds have received attention internationally (Lloyd, 2003). For example, ingestion of plastic debris by albatross chicks is reported to have caused mortality through dehydration, gut blockage and/or toxic effects during digestion (Auman et al., 1998). Such issues are likely to be minimal at well-operated oyster farms.

Adverse effects on seabirds from elevated oyster culture could, however, arise due to the displacement of food sources, although such effects have not been observed in the case of oyster culture or in naturalized populations of introduced Pacific oysters. A study of the ecological role of naturalized Pacific oysters 20 years after their introduction in Argentina study revealed higher densities of local and migratory birds, and higher foraging rates, inside oyster beds compared with reference areas, which were attributed to greater prey availability (Escapa et al., 2004). In the case of elevated intertidal culture, trophic modelling by Leguerrier et al. (2004) similarly suggested that birds could benefit from an enhanced food supply.

Clearly, the consequences for birds and other higher trophic level animals that arise as a result of intertidal oyster farm effects on their food supply (nature, quantity and availability) will depend on their dietary preferences and ability to adapt to changes induced by cultivation. Overall, the few studies of oyster culture effects provide information consistent with other forms of aquaculture, suggesting an attraction of many seabird species to culture areas for foraging on fish and fouling epibiota, and even the cultured crop itself (Ross et al., 2001; Roycroft et al., 2004; Kirk et al., 2007). There has been related discussion of whether the aggregation of bivalve-feeding birds at culture sites may act as a pathway for disease transmission to the culture species and to human consumers (see McKindsey et al., 2006 and references therein); however to our knowledge associated ecological risks have not been identified.

Despite their potential to provide food sources and other habitat (e.g. roosting structures), the large areas of estuarine habitat that may be occupied by intertidal shellfish farms means that they also have the potential to displace seabirds from foraging sites. For example, any bird species that avoid structured habitats may be susceptible to displacement effects. The evidence for such effects is equivocal and indicates that influences will be species and situation-specific (see Dumbauld et al., 2009 and references therein). For some bird species there is evidence of avoidance or a decreased association with oyster structures compared with open tidal flats (e.g. wintering shorebirds in California; Kelly, 2001). In contrast, the few other published studies directly investigating interactions between elevated oyster culture and birds provide little evidence for significant adverse effects. A study of intertidal cultivation in California concluded that off-bottom oyster long-lines did not negatively affect the foraging behaviour of most bird species, but rather enhanced it. In that study there was a greater diversity of birds, and a greater density of some species of shorebird and wading bird, in long-line plots compared with controls (Connolly and Colwell, 2005). In relation to trestle culture in Ireland, Hilgerloh et al. (2001) found that oyster structures did not affect the feeding behaviour of birds. For most species, bird densities were lower in the farm area than a reference area; however, the authors recognized that this pattern may have reflected natural environmental differences. In addition to modifications to benthos, Hilgerloh et al. (2001) also noted that macroalgae fouling of oyster trestles and associated small mobile gastropods provided a food source for some species.

Bird disturbance from noise and traffic does not appear to have been investigated in relation to elevated culture in estuaries, nonetheless it is generally recognized that some seabird species are sensitive to human activities (Goss-Custard and Verboven, 1993; Kelly, 2001; Butler, 2003; Roycroft et al., 2004). For example, Goss-Custard and Verboven (1993) found that oystercatchers were disturbed by the presence of humans in foraging areas, but were also surprisingly flexible in their ability to redistribute their foraging activities. In New Zealand, Butler (2003) found that nesting king shags were highly susceptible to disturbance by boats, leading to part or complete abandonment of nests and chicks.

4.4. Interactions with marine mammals

There are a number of publications concerning interactions between marine mammals and aquaculture (e.g. Würsig and Gailey, 2002; Kemper et al., 2003), but few address intertidal culture of oysters or other bivalves. For aquaculture generally, potential effects on marine mammals include displacement, entanglement, in-water noise, alteration of trophic pathways, and disruption of migration pathways in the case of large cetaceans (Watson-Capps and Mann, 2005). In relation to intertidal cultivation specifically, issues of entanglement or attraction (e.g. to external food inputs) sometimes associated with other forms of aquaculture or fishing do not appear to be regarded as particularly significant (Würsig and Gailey, 2002); marine mammal entanglement appears a greater issue in industries where loose or thin line is used (e.g. Suisted and Neale, 2004).

Watson-Capps and Mann (2005) suggest that small cetaceans may avoid shellfish farms because of human activities, exclusion by structures, or as a result of effects on water clarity or prey availability. These authors report exclusion of bottlenose dolphins (*Tursiops aduncus*) by pearl oyster farms in Western Australia, in a bay where racks were suspended or fixed to the seabed in relatively shallow water (~2–4 m deep). Tracks of individual dolphins showed that adult females tended to stay on the periphery of the farm boundary rather than travel through it. Field and captive studies have found that smaller dolphin species appear reluctant to swim through wooden structures or those with ropes (Kastelein et al., 1995; Watson-Capps and Mann, 2005; Heinrich, 2006). Overall, the nature of habitat exclusion will greatly depend on the type of culture method and the particular species of marine mammal present in the cultivation area. As such, site-specific knowledge is required in order to undertake a robust assessment of risks.

Würsig and Gailey (2002) raise the general issue of potential effects from vessel traffic on acoustic communication, although there appear to be no studies that have specifically addressed such possibilities. In-water noise, especially vessel noise, is regarded as the primary issue of concern because of sound travel in the water column. The potential significance of in-water noise from oyster culture will depend primarily on the vessel traffic generated as a result of oyster farming relative to other activities. As oyster farmers undertake most work during lower tidal conditions when elevated structures are out of the water, in-water noise generation from non-vessel farming activities is likely to be minor. Presumably, any effects on trophic interactions from intensive oyster cultivation (see above) could also have consequences for marine mammal food sources. Again, however, the significance of such interactions is unknown.

4.5. Non-indigenous species and pest organisms

The historic role of the oyster industry in the global spread of non-indigenous species, biofouling pests, toxic or noxious microalgae, and disease is well-recognized. This is especially true in the case of macroscopic biofouling (Boudouresque et al., 1985; Minchin, 2007; Mineur et al., 2007; McKindsey et al., 2007), and associated organisms (e.g. Duggan, 1979; Utting and Spencer, 1992). A number of studies have also documented survival of toxic and nuisance microalgal species (e.g. those associated with biotoxin production and shellfish poisoning) as a result of aquaculture transfers, and oyster transfers in particular (McKindsey et al., 2007). In fact, the introduction of Pacific oysters for aquaculture, and other oyster species to a lesser extent, is regarded as one of the most important historical pathways for the global spread of non-indigenous species (Verlaque, 2001; Wallentinus, 2002; McKindsey et al., 2007). Ruesink et al. (2005), for example, estimated that more than 40% of non-indigenous marine species in Europe, the western United States, and North Sea may have been introduced by oyster aquaculture.

The development and implementation of risk-based guidelines for aquaculture transfers (e.g. ICES, 2005b) are likely to have minimized the present day risk of inadvertent transfers of unwanted species with movements of oysters and seed-stock, especially at international scales (McKindsey et al., 2007). However, at smaller spatial scales there remains the risk that oyster aquaculture activities will lead to the secondary transfer of unwanted organisms, either from previously established populations or from new introductions that have been mediated by other pathways (e.g. global vessel movements). Hence, below we discuss risks to natural ecosystems as a result of oyster cultivation and transfer, considering the transfer of fouling species and diseases separately. The focus of our discussion is on non-indigenous and pest species, but it is worth noting that regional scale shellfish transfers have the potential to extend the range of indigenous species beyond natural barriers to their dispersal, which is arguably an important but overlooked effect (Forrest et al., 2009). Literature on the role of shellfish aquaculture in the introduction of toxic and nuisance phytoplankton tends to focus on species that pose risks to aquaculture operations and human health (e.g. Crawford, 2003). A review of this issue can be found in McKindsey et al. (2007), and is not further discussed here.

4.5.1. Fouling pests

Elevated or suspended structures (and associated shellfish crops) appear to provide ideal environments for some fouling species to proliferate at high densities, which can become problematical in terms of shellfish aquaculture production (Carver et al., 2003; Lane and Willemsen, 2004; Ramsay et al., 2008). Infected structures can also act as reservoirs for the subsequent spread of fouling pests to natural habitats, which can in some instances lead to significant ecological effects. Pest species associated with oyster transfer that may be problematic to oyster cultivation and are also reported to adversely affect natural habitats include various bivalve species (Carlton, 1992), macroalgae such as *Codium fragile* ssp. *tomentosoides*, *Sargassum muticum* and *Undaria pinnatifida* (Trowbridge, 1998, 1999; Verlaque, 2001; Forrest and Taylor, 2002; Britton-Simmons, 2004; Mineur et al., 2007), tunicates such as *Ciona intestinalis*, *Styela clava* and *Didemnum vexillum* (Coutts and Forrest, 2007; Ramsay et al., 2008), and gastropods such as the slipper limpet *Crepidula fornicata* (Goulletquer et al., 2002).

The spread of such species from infested farms at local scales (e.g. within bays) is likely to be primarily driven by natural mechanisms; in particular the dispersal of planktonic propagules in water currents (Forrest et al., 2009). In contrast, spread at inter-regional scales often occurs via inadvertent transport with human activities (Minchin, 2007). For example, infested equipment, seed-stock or crop may be transferred among growing regions as part of routine oyster culture operations (Taylor et al., 2005). There is a high likelihood that associated fouling organisms will survive if such transfers occur without the application of treatments to reduce biosecurity risks (Forrest and Blakemore, 2006; Mineur et al., 2007). In this way, oyster farming activities have the potential to spread marine pests into natural habitats far from the founding population of the pest organism, potentially leading to irreversible effects on natural ecosystems (Ruesink et al., 2005). Hence, there is increasing interest in the development of treatment methods to reduce the spread of fouling pests with regional scale oyster industry transfers. For equipment there are a range of simple options that are straightforward to apply, whereas seed-stock and crop transfers are more problematical in that some treatments effective against fouling may also be detrimental to the culture species (e.g. Forrest and Blakemore, 2006). However, we note that in the case of elevated intertidal culture the tidal height at which the crop is grown can prevent or reduce infection by many of the notorious pests described for subtidal floating systems (Ramsay et al., 2008). In New Zealand, for example, the clubbed tunicate *S. clava* can reach high densities on intertidal rack structures, but is often uncommon at the top of the racks where crop grow-out occurs (B. Forrest, pers. obs.).

In addition to the transfer of non-indigenous fouling organisms, it is important to recognize that oysters cultured in many countries (in particular Pacific oysters) are also a non-indigenous species (Ruesink et al., 2005; McKindsey et al., 2007). Pacific oysters are invasive primarily in artificial structures and in rocky habitats (Ruesink, 2007), but can also invade soft-sediment estuarine habitats (Cognie et al., 2006; Smaal et al., 2009). Naturalized populations of Pacific oysters in their adventive range can reach high densities in the estuaries where they are farmed, as well as in ports and harbours generally. Cognie et al. (2006) found that as much as 70% of the oyster stock in a Pacific oyster growing area of the French Atlantic coast comprised naturalized rather than cultured oysters. Concerns regarding naturalized populations of non-indigenous oysters primarily relate to their ecological impact and effect on amenity values (e.g. Hayward, 1997; Ruesink et al., 2005; Cognie et al., 2006; Diederich, 2006). Based on the many studies cited in this paper highlighting the structural and functional role of oyster reefs in natural ecosystems, it can be expected that dense aggregations of naturalized oysters have the potential to lead to significant ecological changes (arguably both adverse and beneficial) in habitats where they establish (e.g. as described by Escapa et al., 2004 for Pacific oysters in Argentina). However, the ecological role of naturalized populations is likely to be species- and situation-specific. For example, McKindsey et al. (2007) note that non-indigenous *Crassostrea* species in their adventive range do not form high-relief reefs to the same extent as *C. virginica* in its native range, hence are unlikely to provide the same ecosystem services.

4.5.2. Disease

Disease outbreaks have been ascribed to oyster introductions or translocations, although these appear highly species-specific (see Carnegie, 2005 for a review of effects in culture). The Pacific oyster is by far the dominant farmed oyster, with production of the next most important species (*C. virginica*) being less by an order of magnitude (FAO, 2006b,c). Other minor species noted by Garibaldi (1996) are (in order of importance) *Crassostrea iredalei*, *Saccostrea commercialis*, *Ostrea edulis* and *C. rhizoporae*. In addition, *Crassostrea ariakensis* (Cochennec et al., 1998) and *C. sikamea* (Ruesink et al., 2005) are worthy of mention in the context of disease risk.

Diggles et al. (2002) reports several diseases and parasites associated with Pacific oysters, most of which are globally ubiquitous and appear to pose a threat to oyster production (especially in hatcheries) or product value rather than natural ecosystems. These include various species of planocerid flatworm and mud-worm (Handley and Bergquist, 1997; Handley, 2002) and the ostreid herpes virus (OsHV-1) (Hine et al., 1992). Summer mortalities of Pacific oyster seed have been linked, but so far inconclusively, to this virus in California (Friedman et al., 2005). Other pathogens implicated in summer mortality include *Vibrio* spp. (Bower, 2002) and *Nocardia crassostreae* (Bower, 2006a). A variety of other diseases have been reported in Pacific oyster culture, including oyster velar virus disease which resulted in mortalities approaching 100% (Bower, 2001a), ciliate infections associated with mortalities exceeding 50% in oyster seed (Bower, 2001b), and *Marteilioides chungmuensis*, which degrades the appearance of Pacific oysters and thereby reduces marketability (Bower et al., 2006).

In New Zealand, pathogens of Pacific oysters have been extensively investigated and no organisms listed by the World Organisation for Animal Health (OIE, 2001) or other significant pathogens have been reported (Hine, 1997; Diggles et al., 2002; Hine, 2002). Culture of Pacific oysters in New Zealand is considered unlikely to pose a threat to naturalized conspecifics or other species. Elsewhere, however, Pacific oysters have carried pathogens following human transportation. For example, nocardiosis, reported from Pacific oysters and *O. edulis* (Bower, 2006b) is caused by the bacterium *Nocardia crassostreae* which is thought to have originated in Japan and then spread to the west coast of North America with Pacific oysters (Straus et al., 2008).

The Pacific oyster appears to be more resilient to the significant diseases (Elston, 1993; FAO, 2006a) suffered by other oysters. For instance, it is partially resistant to *Perkinsus marinus* (Bower, 2006b) which ravages *C. virginica*. Similarly, in comparative challenges with *Mikrocytos mackini* (Bower, 2007a), Pacific oysters appear more resistant than other oysters, and *Haplosporidium nelsoni* appears to have greater pathological impact on *C. virginica* than *C. gigas* (Bower, 2007b). The decline of *C. virginica* in the eastern United States due to over-harvesting (Mackenzie, 1996) has also been in part attributed to *H. nelsoni* and *P. marinus* that may have been introduced with Pacific oysters (Carnegie, 2005). The protozoan parasite *Marteilia refringens*, although thought to have occurred in Pacific oysters, has been confirmed in oysters such as *C. virginica*, *O. edulis*, *O. chilensis*, and non-ostreid bivalves (Bower, 2007c).

The apparent advantage to aquaculture of Pacific oysters being relatively disease-resistant also presents a liability in that this species potentially provides an asymptomatic reservoir of pathogens that could be more damaging to other oysters and bivalves. Thus, its higher resistance might offer advantage or disadvantage depending on context. It is apparent that even where disease in oysters is reasonably well known, the likelihood and consequences (e.g. enhanced spread or virulence) of disease transmission from cultured stock to natural ecosystems is incompletely understood to the extent that a precautionary approach is warranted. Although Pacific oysters appear relatively disease-resistant, this species potentially carries a number of virulent pathogens that should be screened out before stock is moved. Safeguards are also justified for minor oyster species. *C. ariakensis*, for example, can show resistance to *P. marinus* (Powers, 2006), while *C. ariakensis* from its home range (East Asia) has harboured other *Perkinsus* species as well as three strains of herpes virus and several other pathogens (Moss et al., 2007) that should be excluded from any population intended for translocation. Given evidence that *C. gigas* may hybridize with other *Crassostrea* species (e.g. Huvet et al., 2004; Powers, 2006), consideration of changed disease susceptibility through hybridization is warranted. Depressed performance of *M. edulis* × *M. galloprovincialis* hybrids is documented by Beaumont et al. (2004), and Fuentes et al. (2002) report their lower hybrid viability when challenged by infection with *M. refringens*.

5. Synthesis of ecological effects and evaluation of relative risks

5.1. Rationale and approach

To inform further research and management, it is useful to consider the relative importance of the different ecological effects of elevated oyster aquaculture. The benthic impacts of elevated culture are well documented by comparison with many of the other interactions between cultivation and the environment. Nonetheless, from the information presented in this paper it is evident that the broad range of key ecological effects is sufficiently recognized that the potential for adverse consequences can be evaluated in a relative context. The qualitative risk ranking method that is widely used in risk assessment (Burgman, 2005), and which has previously been used to assess aquaculture effects (Crawford, 2003), provides a useful screening tool for understanding relative risk. The method involves applying subjective scores to both the relative likelihood that a particular effect will occur, and the magnitude of its consequences.

The judgement of consequences in ecological risk assessment often includes notions of the severity, extent and reversibility of effects (e.g. Suter, 1990; Crawford, 2003; Serveiss et al., 2004). These three criteria were made explicit by Emmett (2002) in relation to shellfish aquaculture, and are adopted in the present paper (Table 1). Hence, for each of the issues identified for elevated oyster culture (i.e. reflected in the section sub-headings throughout the paper), we evaluated relative ecological risk as low, medium or high in relation to: (i) the severity of adverse effects, without consideration of their

Table 1
Categories and scores used to assess the relative ecological significance of effects from elevated oyster culture, and level of knowledge on which the assessment is made.

Consequence category	Consequence score		
	Low	Medium	High
Severity of effect[a]	Minor	Moderate	Major
Spatial extent of effect	Local scale (<100 m from culture structures)	Bay-wide (100 m–1 km from culture structures)	Regional (>1 km from culture structures)
Duration of effect	Short-term (abates within <1 year)	Medium term (continues for 1–5 years)	Long term (continues for >5 years and may be irreversible)
Knowledge base	Based on perception or inference from related studies	Based on limited information on effects of elevated culture	Specific effects of elevated culture well known

[a] Severity was assessed according to criteria previously used for shellfish aquaculture, and by discussion and consensus among the assessors (see text).

spatial extent or duration; (ii) the spatial extent of effects from site-specific to regional scales; and (iii) the duration of impact, in terms of the length of time effects would persist if farming operations were ceased and structures removed (Table 1). Issues were scored independently even though they may be inter-related; for example the effects of habitat creation were scored separately from (and did not include) the effects of fouling pests. Severity was assessed in relation to qualitative criteria used elsewhere for shellfish aquaculture (Emmett, 2002; Crawford, 2003) and considered the potential for adverse ecological changes to populations, communities and ecosystems in terms of structure (e.g. abundance, diversity, dominance) or function. The likelihood of adverse effects was scored in five categories (highly unlikely, unlikely, possible, probable, almost certain). The potential for the greatest adverse effect in a relative sense arises where there is a very high likelihood of a major ecological change that is irreversible and widespread. Note that as the focus of the assessment was on adverse effects, any effects that were regarded as positive or beneficial were scored as low risk (i.e. low likelihood and consequences of an adverse impact).

Scores against the consequence and likelihood criteria were made by three of the paper authors (Forrest, Keeley, Hopkins) based on the information herein and their individual experience. Although some correlation in assessor views was expected, for the purpose of providing guidance on relative risk we did not consider that canvassing a broader expert group was necessary. Obviously, actual levels of risk for most of the issues described in this paper will be context and scale-dependent, for example relating to site-specific factors such as the intensity of oyster farming in a given area, the sensitivity of the receiving environment, the presence of pre-existing stressors, and the extent to which mitigation of any adverse effects is undertaken. Hence, we provide risk intervals as a measure of the variability or uncertainty regarding effects (Burgman, 2005), rather than point estimates of risk as previously undertaken for shellfish aquaculture (e.g. Crawford, 2003). We recognize that it is desirable to have measureable endpoint criteria for the narrative categories of severity in Table 1 (Suter, 1990); however, we note that no clear and unambiguous standards for ecological change exist. Hence, to reduce discrepancies in scoring due to linguistic uncertainty and ambiguity with respect to the categories used (Regan et al., 2002), the three assessors independently assigned their range of scores, discussed reasons for differences, and reached a final range by consensus.

The five categorical scores of likelihood were converted to values from 1 to 5 (highly unlikely to almost certain), while severity, spatial extent and duration were scored as 1 (low), 2 (medium) or 3 (high). The calculation of consequences was treated as additive across these three categories ($severity + extent + duration$), and risk calculated as $likelihood \times consequences$, with a maximum possible score of 45. For each issue, interval arithmetic was used to calculate the risk interval as the range from minimum to maximum values scored by the three assessors. We make no attempt to rank risks for each issue in terms of their acceptability, as this is inherently a value judgement that will differ from person to person and among different cultures. Furthermore, we emphasise that our purpose here is to provide a screening tool for ascertaining the risk of adverse effects for the different issues *relative* to each other, and to identify areas of potential concern for which greater understanding or quantification may be desirable.

5.2. Key findings

Perhaps the most interesting point that arises from the risk assessment (Fig. 3A) is that the role of elevated oyster culture in the spread of pest organisms emerges as being particularly significant. This finding is consistent with an aquaculture risk assessment described by Crawford (2003) for Tasmania, and also with the general view that inadvertent pest introduction is one of the more significant issues associated with aquaculture in estuaries (DeFur and Rader, 1995). The reason is that, by comparison with all other issues, the spread of pest organisms by oyster farming can occur at regional scales (e.g. as a result on seed-stock transfer) potentially leading to ecologically significant and irreversible changes to coastal ecosystems (Elliot, 2003). Although, management approaches may be developed to minimize any pest risks that are considered unacceptable (e.g. treatment of seed-stock before regional transfer), there are few examples where such strategies have been completely effective (Piola et al., 2009). Furthermore, the non-target effects of control methods may also need to be considered (e.g. Dumbauld et al., 2006; Mallet et al., 2006; Locke et al., 2009). By comparison with pest organisms, the lower score but relatively wide risk interval for disease reflects the fact that this is not generally a significant issue for natural populations or ecosystems in the case of the dominant culture species (Pacific oysters), but may be important for other cultivation species (Fig. 3A).

Despite the benthic effects of elevated oyster culture being relatively well-studied (Fig. 3B), the potential for adverse effects was judged as intermediate among the range of scores that were assigned across the different issues (Fig. 3A). While the effects of biodeposition and farm debris can be among the more obvious effects of oyster farms (where the latter is unmanaged), the ecological implications are relatively localized. Although some benthic effects (e.g. organic enrichment) may abate over time scales of several months to a few years, accumulated shell and debris could (unless removed) lead to long-term changes in estuarine habitat structure. The effects of contaminants, adverse effects on water quality and effects on marine mammals had the lowest risk scores (Fig. 3A), as the likelihood of adverse effects was considered very low even though the knowledge base was minimal (Fig. 3B). However, in the case of marine mammals we recognized that a very low likelihood interaction could have significant consequences if critical habitat or endangered species were adversely affected. Risks arising from inter-related issues of altered nutrient cycling and SPM received intermediate risk scores but relatively wide intervals, indicating that effects can be more pronounced where large scale intensive cultivation occurs (Fig. 3A,B). Crawford (2003) expressed the view that effects on other filter-feeders as a result of food depletion by oyster cultivation were unlikely, as oyster farmers had an economic incentive to keep stocking densities below carrying capacity.

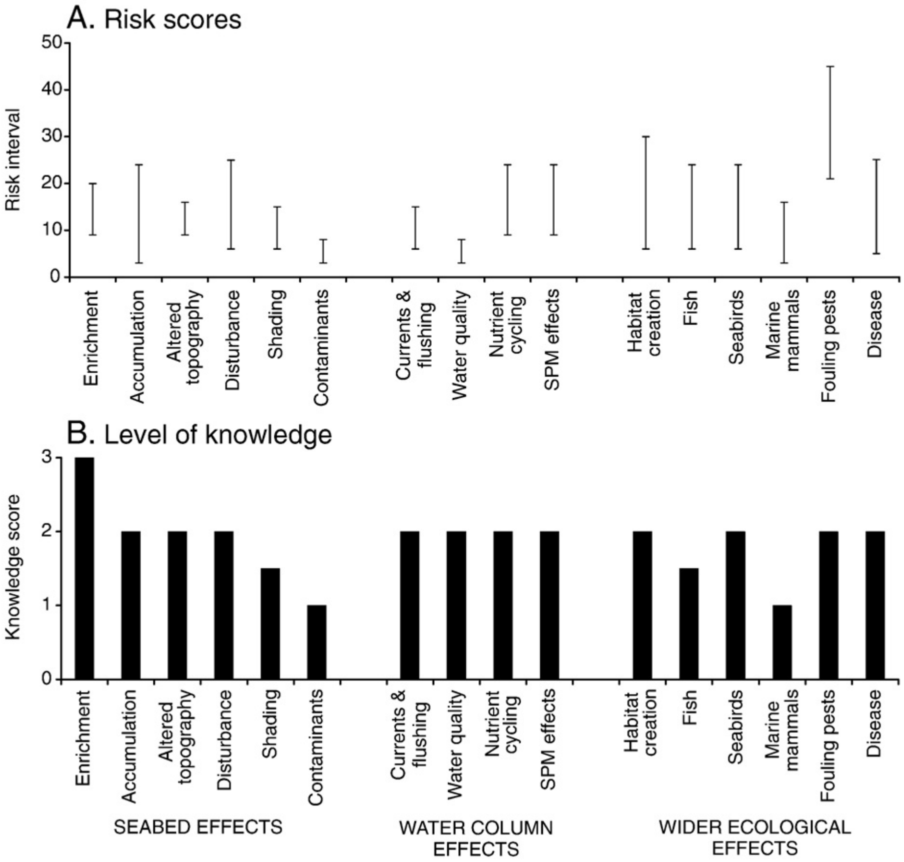

Fig. 3. Summary of: A. relative risk intervals; and B. knowledge scores, based on the consensus opinion of three assessors, for the range of ecological issues discussed in this paper.

The wide interval scored for habitat creation reflected the consensus that the well-recognized effects associated with aquaculture (e.g. local increase in biodiversity, provision of fish habitat) can be regarded as beneficial (reflected by the low end of the risk interval), yet the functional role of extensive areas of artificial habitat is poorly understood but has the potential to lead to adverse consequences (reflected as the high end of the risk interval). As an example of the latter, the assessors discussed a general scenario in which artificial habitat enhanced recruitment of an important predator with potential to cause cascading effects in adjacent natural ecosystems (e.g. described for floating piers in the case of the common jellyfish *Aurelia aurita*; Miyake et al., 2002). Where the effect of habitat creation on fish and seabirds has been specifically considered there is little evidence of significant adverse effects, but recognition that there is potential for such effects depending on species and context.

Overall, with the possible exception of pest species, the generally low to intermediate risk scores assigned across the issues largely concurs with the review of Dumbauld et al. (2009) who concluded that bivalve culture effects in US West Coast estuaries tended to be primarily localized and short-term, and not associated with larger scale ecosystem changes. In relation to shellfish aquaculture in New Zealand, Keeley et al. (2009) suggested that societal views on acceptability were likely to constrain development to a level that minimized the risk of significant ecosystem-wide effects. Nonetheless, it should be acknowledged that unrecognized estuary-wide or cumulative effects could have already occurred from some oyster farm developments, or could arise, for example: (i) in situations of high intensity oyster farming (e.g. in enclosed embayments dominated by oyster farms), or (ii) because of the occurrence of baseline ecological values of high importance. Without a comprehensive knowledge of baseline conditions and subsequent changes post-farm development, many of the wider or ecosystem-level impacts described in this paper would be difficult to clearly determine retrospectively. Some uncertainty regarding effects is inherent in the wide risk intervals in Fig. 3A. For many of the issues the width of these risk intervals also reflects scale or context dependence. Wide intervals are inevitable when assessing the issues in a general context as in this paper, but may be reduced by evaluating specific culture sites and scenarios. Similarly, risk intervals may be reduced by disaggregating the issues (Burgman, 2005), for example by separately evaluating the positive and negative effects of habitat creation.

6. Conclusions and future directions

It is evident from recent reviews that the research focus and state of knowledge for elevated oyster culture described in this paper largely parallels that for other cultured bivalve species (and to some extent finfish) and other cultivation methods (e.g. McKindsey et al., 2006; Forrest et al., 2007b; Dumbauld et al., 2009; Keeley et al., 2009). Whereas the severity of some effects (e.g. biodeposition) may be quantitatively greater for other species or methods, there is considerable overlap in the ecological issues. As evident in this paper, although the general effects of elevated oyster culture are known and their relative ecological significance can be evaluated, there are still knowledge gaps and areas of uncertainty. The focus of previous research on assessment of benthic impacts (and phytoplankton depletion to a lesser extent) has resulted in other issues, which are arguably more important or less easy to manage, being overlooked. The association between oyster culture and the secondary spread of pest species, which could have significant non-local and irreversible consequences, is a case in point. Clearly, there is a need to redress the balance of effort in future studies. This could include, for example, site-specific risk profiling for actual and potential pests (e.g. assessment of the likelihood that high risk pest species will establish), estimation of the significance of pest spread by oyster farming pathways relative to other sources of risk (e.g. vessels), and consideration of the feasibility of management (Taylor et al., 2005; Forrest et al., 2006).

Even though effects on the seabed are comparatively well understood and recognized for most types of aquaculture, there are nonetheless an overwhelming number of descriptive studies. There is still a need for greater application of model-based approaches to understand and predict the magnitude of effects as a function of key environmental (e.g. flushing characteristics) and farming-related (e.g. stocking levels, farm size and age, farming method) factors (e.g. DEPOMOD; Cromey et al., 2002; Weise et al., 2009). Similarly, there is scope for manipulative experimental approaches to elucidate for these same factors the relative importance of the key mechanisms that lead to benthic effects (e.g. sedimentation, enrichment, physical disturbance, shading, habitat creation). Acquisition of such knowledge would provide guidance for managers by identifying the types of environments or practices that allow oyster farming to be carried out with minimal impact, especially in relation to direct habitat change beneath cultures.

Research to address many of the complex issues where information gaps are evident (e.g. water column effects, functional changes, effects on higher trophic level animals) will require greater understanding of ecosystem processes, many of which occur beyond the immediate environment of the cultivation area (e.g. changes to food-web pathways). While modelling and related approaches have been undertaken to evaluate trophic effects from culturing oysters (Leguerrier et al., 2004; Lin et al., 2009) and other forms of bivalve aquaculture (e.g. Jiang and Gibbs, 2005; Anderson et al., 2006), the large amount of data required for reliable model estimates may limit their general utility outside specific case study areas. Hence, progress with understanding some of these complex issues will probably be slow, as it will require fundamental coastal ecosystem research in a range of environments. In the meantime, it is apparent that although there is some evidence of bay-wide ecological changes as a result of intensive intertidal shellfish cultivation, there appear to have been no catastrophic consequences.

Furthermore, primarily as a result of societal expectations, regulators and industry are increasingly managing or mitigating the potential for adverse effects from aquaculture in a comprehensive manner that addresses the range of ecological risks described in this paper, for example through the development of codes of practice and management plans (e.g. EPA, 2005; Taylor et al., 2005). Arguably, cultivation effects should be considered from an even broader perspective that recognizes the ecological changes resulting from cultivation collectively. When the range of effects is considered as a whole it could be argued that some nominally adverse effects may be compensated to some extent by more positive effects. For example, although natural seabed sediments and benthos may be altered beneath cultivation structures, benthic production may increase. Together with the creation of novel habitat, such changes may benefit some fish and bird species and provide a range of other beneficial ecosystem services such as local enhancement of biodiversity. More broadly, we suggest that management planning and responses to the development of oyster and other shellfish aquaculture in estuaries should be made in relation to other sources of environmental risk, and recognize the cumulative effects of anthropogenic activities in estuarine systems at a bay-wide or regional scale (e.g. Anderson et al., 2006), so that the effects of aquaculture are placed in context.

Acknowledgments

We are grateful to Richard Piola and three anonymous referees for helpful review comments on a draft of this manuscript. Many thanks also to Paul Gillespie, Kim Clarke, Bethany Roberts and Robyn Dunmore for their input to earlier drafts and the risk assessment exercise. This work was partly funded by the New Zealand Foundation for Research, Science and Technology Envirolink scheme.

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Fisheries and Aquaculture Journal

Gallardi et al., Fish Aquac J 2014, 5:3
DOI: 10.4172/2150-3508.1000105

Review Article Open Access

Effects of Bivalve Aquaculture on the Environment and Their Possible Mitigation: A Review

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Abstract

Bivalve aquaculture, in particular oyster, clam, scallop and mussel culture, is a globally increasing activity. Increased bivalve production translates inevitably into increased impact on the environment surrounding the aquaculture activities. The effects of this type of aquaculture on the environment are often considered less important compared to those of finfish culture. However, bivalves due to their natural characteristics are considered keystone species in the ecosystem and therefore they have the ability to affect the surrounding environment in both negative and positive ways. They influence primary and secondary productivity and start a series of cascade effects on water column and sediment population and dynamics. The purpose of this article is to present a review of the effects of bivalve aquaculture on the surrounding environment and the current mitigation strategies. In addition, this review highlights how the same natural characteristics of bivalves can positively interact with the environment, and the possible use of bivalve aquaculture as restoration and remediation tool for marine environments.

Keywords: Bivalve; Aquaculture; Environment interactions; Benthic habitat; Positive effects; Sustainability

Introduction

Global bivalve aquaculture production has been increasing constantly over the past 20 years. The main bivalve species cultured in the world are oyster, clam, scallop and mussel [1]. Bivalve aquaculture is considered to have less dramatic environmental impact compared to finfish culture, since it requires minimal addition to the environment. For the main species cultured the food is supplied by the environment itself and the wastes return nutrients and minerals to the ecosystem [2]. However, bivalves have the ability to maintain, modify and create entire habitats due to their effects on suspended particles and their shell formations [2,3]. The majority of research on interaction between bivalve culture and the environment has been concentrated on mussel and oyster, probably because the production of other bivalves such as clams and scallops is greater in Asia, where aquaculture and its effects have been a culturally accepted part of the coastal environment for centuries [4].

The purpose of this article is to present a review of the effects of bivalve aquaculture on the surrounding environment, particularly in estuarine and coastal zones, the current prevention and mitigation strategies and highlight how bivalve culture can positively interact with the environment.

Bivalves are suspension feeders that perform their functions in a range of habitats, in particular estuaries, lagoons and coastal oceanic systems. They gain nourishment by filtering suspended particles such as phytoplankton and detritus from the water column [3]; it has been calculated that an oyster can filter on average 15 to 55 liter/day of seawater [5,6]. Bivalve by-products are dissolved ammonium and bio-deposits of feces and pseudofeces; they sequester nitrogen in the form of protein in meat and shell and stabilize phytoplankton growth dynamics through the moderation of ammonia cycling in the water column. They are therefore considered "keystone" species which exert "top-down" control of phytoplankton by grazing but also "bottom-up" control through biodeposition and promotion of nutrient removal [3,6-9]. Epifaunal bivalves such as oysters and mussels respond to increased levels of phytoplankton and detritus in the water column with increased filtration capacity and production of pseudofeces. Infaunal bivalves such as clams adjust their clearance rates rather than increasing production of pseudofeces [9,10]. These processes affect the food web, the biogeochemical cycling, and the physical and chemical environment, potentially modifying habitats and ecological functioning [3,9].

Bivalve aquaculture has therefore the ability to affect the environment in both negative and positive ways, with a variety of near and far field cascading effects on different parts of the ecosystem, including influencing primary and secondary productivity and community structure. Culture structures and operations can alter water flows, sediment composition and sedimentation rate, and disturb the benthic flora and other marine organisms [4,9,11,12].

Estuaries are often a preferred site for bivalve culture and therefore are the environments examined by the majority of literature. The species that can exert the highest influence on the environment are oysters and mussels, since they maintain high clearance rates and reject large numbers of particles as pseudofeces [2,8]. The direct comparison of impact between clam (*Tapes philippinarum*) and mussel (*Mytilus galloprovincialis*) culture located in the same body of water has in fact shown a greater impact for mussel culture [13].

There is more than one way a system can react to bivalve culture:

- The bivalve can redirect energy away from the zooplankton, replacing its ecological role and grazing on phytoplankton, leading to less energy passing up to the higher pelagic trophic level.
- Bivalves can direct energy away from benthic filter-feeders,

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Received May 30, 2014; **Accepted** August 28, 2014; **Published** September 05, 2014

Citation: Gallardi D (2014) Effects of Bivalve Aquaculture on the Environment and Their Possible Mitigation: A Review. Fish Aquac J 5: 105. doi: 10.4172/2150-3508.1000105

Citation: Gallardi D (2014) Effects of Bivalve Aquaculture on the Environment and Their Possible Mitigation: A Review. Fish Aquac J 5: 105. doi: 10.4172/2150-3508.1000105

decreasing their population and opening an ecological niche for benthic deposit feeders.

- Finally, bivalves can redirect energy from the bacteria in the microbial loop by recycling particulate nitrogen to inorganic nitrogen resulting in a noticeable increase in rates of primary productivity and less detritus.

It is however likely that these three processes can occur concurrently [14]. Based on Gavine & Mc Kinnon's [15] hazard assessment for oyster (Pacific, Sydney rock and pearl) and blue mussel (*Mytilus edulis*) culture in Australia, the higher risk of impact was associated with the deterioration of sediment quality, the alteration of sediment physical structure and the impact on seagrass beds.

The effects of bivalve aquaculture on the environment have been classified in different ways throughout the literature. For the purpose of this manuscript they will be divided in four main effects: on the water column, on the sediment, effects of bivalve introduction and effects on other marine species (including marine mammals and birds). A summary of these effects is reported in Table 1.

Ecological Effects of Bivalve Aquaculture

Effects on water column and nutrients

Bivalves affect the water column through filtration and grazing, and through modification of the nutrient cycle, with direct excretion and microbially mediated remineralisation of their organic deposits in sediments [9,10]. Large bivalve assemblages have the ability to modify phytoplankton populations and blooms. They can regulate the abundance of phytoplankton in shallow seas and reduce phytoplankton bloom intensity. Reduced turbidity due to bivalve grazing can increase light, a limiting factor for the growth of other species, such as algae [2,8,10,16,17]. Grazing reinforces seasonal successional cycles in phytoplankton composition. Picoplankton is favored by warmer waters and by changes in relative abundance of organic and inorganic nitrogen and it is also retained less efficiently on the gills of bivalves, while nanoplankton is preferentially removed by grazing. Therefore, during warmer seasons, with the help of bivalves, picoplankton becomes relatively more abundant than larger species [8,18,19].

The effects of bivalves on nutrient cycling include marked changes in the nitrogen distribution, especially contribution of nitrogen in the form of ammonium (NH_4^+), removal of phosphorus through biodeposition and recycling of silicate through transfer from water column to the sediment. The ammonium excreted by bivalves is immediately available for primary production; therefore bivalves have a positive effect on primary production by increasing the nitrogen turnover in the water column. Bivalves such as mussels may also concentrate certain metals like copper in their pseudofeces [2,3,16]. The extent to which the overall nutrient budget and primary production are affected by bivalves is related to their abundance, location, system flushing rate and residence time. Therefore, a population of cultured bivalves has the potential to modify the nutrient cycle in coastal ecosystems in that carbon and nitrogen ingested as phytoplankton are converted into other forms and concentrated near the culture area. It has been postulated that changes in relative concentration of silica, nitrogen and phosphorus could facilitate growth of harmful phytoplankton classes. For instance, promotion of algal blooms of *Pseudo-nitzschia* in relation to eutrophication has been demonstrated, though a direct link between these blooms and bivalve culture sites is still speculation. In the end bivalve aquaculture is a net remover of nutrients from the ecosystem through harvesting of the product [2,3,9,15,16,18].

Effects on sediments and benthic habitat

Bivalve filter-feeders effectively remove natural suspended matter with a diameter between 1 to 7 µm, depending on species, and return large fecal pellets of 500-3000 µm. This pellet rapidly settles to the seabed, particularly when slow or poor water flushing and exchange conditions exist. This particle repacking diverts primary production and energy from planktonic to benthic food webs [2,3]. The bulk of research regarding bivalve aquaculture and sediment focuses on the effects of increased organic load to the sediments from biodeposition, habitat modification associated with culture gear and consequent changes in local fauna [9,10].

In the sediment, the rate of accumulation or dispersion of biodeposits and the severity of impact created by bivalves depends on water depth and prevailing currents close to the seafloor. In the literature, a variety of observations on cultured bivalve biodeposition have been reported,

	Effect	**Consequences**
Water column and nutrients	Phytoplankton modification	Bloom modification
	Reduced turbidity	Increased light penetration
	Increased NH4+	Increased primary production
	Metals concentration	
Sediment and benthic habitat	Increased deposition	Anaerobic sediment
		Increased bacteria and meiofauna
		Decreased suspension-feeders
		Increased deposit feeders
	Modification of topography and hydrography	Habitat creation/modification
	Removal of calcium carbonate	Increased acidification
		Decreased positive feed-back
Other marine species	Nutrient and habitat modification	Increased crustaceans & some fish
		Seagrass displacement
		Disturbance for mammals and birds
		Creation of new habitat for birds
	Food competition	Decreased zooplankton & larval fish
Introduction	Introduction of nonnative species	Diseases introduction
		Pest introduction

Table 1: Main effects of bivalve aquaculture on the environment and their direct consequences. Grey highlight represents effects that can be considered both negative and positive depending on the situation.

Citation: Gallardi D (2014) Effects of Bivalve Aquaculture on the Environment and Their Possible Mitigation: A Review. Fish Aquac J 5: 105. doi: 10.4172/2150-3508.1000105

Effect	Evaluation measure	Prevention	
Phytoplankton modification	Environmental indicators (water)	Ecological carrying capacity models	Ecosystem based management for bivalve aquaculture
Nutrient modification	Environmental indicators (water)	Ecological carrying capacity models	
Increased deposition	Environmental indicators (sediment)	Ecological carrying capacity models	
Benthic fauna modification	Environmental indicators (sediment)	Ecological carrying capacity models	
Habitat modification	Environmental indicators (sediment)	Ecological carrying capacity models	
Effects on marine mammals and birds		Environmental risk assessment BMP & codes of conduct	
Introduction of nonnative species, diseases and pests	Quarantine Hatchery Testing	Environmental risk assessment BMP & codes of conduct Legislation	

Table 2: Evaluation measures and prevention methods for the main effects of bivalve aquaculture on the environment. Evaluation and prevention measures may be employed separately or in conjunction under an ecosystem-based management plan.

including little or negligible impact, low sedimentation rates and absence of major changes in benthic infauna [20-22]. In other cases the benthic community presented strong long-term effects. For example it has been shown that adult mussels (*Mytilus edulis*) are able to increase natural sedimentation rates by an average factor of 26 [2]. Kaspar [16] noted consistently higher nitrogen pools in sediments under a mussel farm, suggesting accumulation of inorganic nitrogen. When the organic biodeposition reaches high levels, its decomposition can increase oxygen demand and generate an anaerobic environment that promotes ammonification, sulfate reduction and silicate flux increase [2,3,8,10,18,23]. Benthic responses to organic enrichment include an increase in bacterial abundance, meiofauna community and biomass, and reduction in macrobenthic infaunal abundance and diversity [2,16,20,23]. Bivalves have the capability to transform a diverse benthic community dominated by suspension feeders (bivalves, crustaceans and some polychaetes) into one dominated by smaller opportunistic deposit feeders, such as polychaetes, scavengers, carnivores and hydrogen sulphide-tolerant species [12,16,19,23-26].

Despite the variety of impact severity found in literature, there is however a consensus on the fact that aquaculture activities in intertidal high energy well flushed areas generally results in dispersal of the organic biodeposits with a lesser impact on sediments. On the other hand, culture in sub-tidal quiescent low energy areas can potentially produce a large accumulation of biodeposits and therefore have a greater localized impact on the benthos [8,9,15,19-21,24,26]. In addition to the physical and hydrodynamic characteristics of the site, the scale of the operation and the technique used will determine the impact of bivalve culture on sediments and benthic populations. For instance, activities such as mussel harvesting by bottom dredging practiced in Germany, Netherlands, Irish Sea and Maine results in greater impacts on benthic habitat [9].

It is important to mention the processes related to bivalve shell formation, which capture carbon in the form of calcium carbonate. After natural mortality of the bivalve, the carbon can be sequestrated in the surface sediment, where it provides local buffering against ocean acidification. Moreover, there is a positive feedback process between bivalves and carbonate addition to the sediment: the bivalve carbonate producers provide a critical sedimentary constituent promoting the long-term survival of their own species [9]. However, with current aquaculture practices the source of carbonate is extracted from the marine system and discarded on land. This practice coupled with ocean acidification can accelerate carbonate loss in estuarine and coastal systems and diminish the positive feedback and the provision of habitat that support recruitment, growth and survival of the bivalves themselves [9]. Bivalve shells also have the potential to change seabed topography and hydrodynamic conditions and therefore provide novel habitats which would normally not occur in a certain environment [10,18,25,26]. It has been indicated that a mussel farm situated in a sheltered site can add up to 10cm/year of biodeposit largely constituted by shells, resulting in changes to the seabed up to 20 m from the farm boundaries [24,26]. The material accumulation can provide sites of attachment for large epibiota, such as tunicates, sponges and calcareous polychaete [16]. Oyster is an important species to consider when discussing shell deposition since oyster reefs are known to provide a habitat that can support a diversity of taxa. When oyster culture is located on soft- sediment habitats, dominated by flat sand or mud, its impact could result in fundamental long-term shifts in benthic community composition [3,10,25]. It has been observed that both oyster and mussel introduction in soft-sediment areas generates an increase in diversity and abundance of infauna and epifauna. Therefore, the establishment of a bivalve culture operation has the ability to shift soft-sediment to hard-bottom, where communities are generally more diverse, have greater biomass and are more productive [4,11]. In addition, aquaculture structures such as bags, anchors and ropes can both change the hydrodynamic processes of an area, redirecting water flow, and alter the benthic habitat [9,27].

The effects on the sediments caused by in-bottom clam aquaculture are not as extensively studied as those of oyster and mussel. However, this type of aquaculture deserves a mention, since it involves a number of practices which largely modify the physical environment, clearing intertidal and beach zones of rocks, wood debris and competing species (unwanted species of clam, mussels and barnacles). Also predator species such as snails and starfish are removed and gravel may be added to encourage growth and stabilize the sediments. The culture is then covered with anti-predator netting, which can facilitate the growth of other species otherwise not suitable for coastal areas and can also trap fish. The removal of rocks and debris has the opposite effect of netting on hard-bottom species, since their main natural attachment sites are removed [27].

Effects on native pelagic and benthic species

Cultured bivalves affect the planktonic and benthic food web by modifying, repacking and increasing the sedimentation rate of fine suspended particles, ultimately altering the availability of food resources to other species. The physical structure of the farm and the fouling that concentrates on bivalves and structures create an attraction for a variety of species. Crabs, other crustaceans, shellfish and demersal fish seem to benefit from culture activities as a result of increased food availability under bivalve suspended culture. In some cases it has been

Citation: Gallardi D (2014) Effects of Bivalve Aquaculture on the Environment and Their Possible Mitigation: A Review. Fish Aquac J 5: 105. doi: 10.4172/2150-3508.1000105

shown that the diet of crabs in vicinity of farms switched from algae to mussels as the main component [2,4,10,15]. Lobsters (*Homarus americanus*) appear to be attracted by both the presence of anchor blocks and other structural components of mussel farms, used as refuge, and by the increased food supply constituted by the bivalves themselves and by other species attracted by the farm [4,28]. It has been observed that kelp (*Laminaria longicrursis*) grows abundantly on cultured blue mussel (*Mytilus edulis*) lines, with new tissue growth both in summer and winter, suggesting that this plant is taking advantage of nutrient release from the bivalves [10]. Bivalve aquaculture can displace or disturb seagrass with culture structures and operations, the farm can also provide an unnatural hard substrate, physical modification of flows and sediments, and shading from light, which affects growth and survival of both macro-algae and seagrass. Moreover, ground-cultured oysters can affect seagrass also by severing the plants with the sharp tips of their shells [9,15,25]. Carnivorous fish can be attracted in areas with bivalve farms by an increase in benthic herbivorous fauna; this increase is due to a rise in microphytobenthos, an important food source for the benthic herbivorous fauna, which is in turn caused by increased water clarity from bivalve filtration [8].

In contrast, zooplankton and larval fish that depend on suspended seston as food can compete with bivalve for grazing. Also, both *Mercenaria* and *Mytilus* species have the ability to significantly reduce abundance of microzooplankton and mesozooplankton through filtering while oysters may have the capacity to filter and remove larvae of some invertebrate species. However, the importance of this last impact under natural conditions is still unknown [2,9,10,19]. Lastly it is significant to note that certain aquaculture practices, such as collection of wild bivalve seed, can have a potential negative impact on native stocks of the cultured species, if collection is done on large proportions of the stocks [11,15].

Introduction of Nonnative Species

In order to diversify the number of species used in culture operations, introduction of nonnative species has been largely employed in aquaculture. The most renowned examples are the introduction of Pacific oyster (*Crassostrea gigas*) on the Pacific and Atlantic coast of North America, in Europe, Australia and New Zealand, and the introduction of Mediterranean mussel (*Mytilus galloprovincialis*) in South Africa [11]. Nonnative bivalve species often exhibit faster growth rates, better resilience to diseases, physiological stress and reproductive output than equivalent native species. Therefore they can become a superior competitor for resources, with the risk of naturalizing and establishing self-sustained populations and dominate endemic species [9,11]. They can influence biodiversity, local community composition and the performance of the whole ecosystem and they are more likely to have negative far-field effects compared to cultured endemic bivalve species. However, there appears to be a lack of knowledge on how oysters and other nonnative bivalves impact community and ecosystem level structure and function [9,11].

Diseases and pest introduction

Numerous diseases have been transferred via movement of infected bivalve stocks. In many cases the fact that the translocated bivalves harbored a disease agent was unknown due to lack of basic knowledge of the disease or inadequate testing and monitoring before the transfer. This has been the case for different oyster diseases. For example, *Haplosporidium nelsoni*, the causative agent of MSX disease, is a parasite that infects Pacific oysters causing little disease and mortality while it greatly affects eastern oysters. Its transfer via movement of Pacific oysters has been the cause of a major decline in eastern oyster population in Chesapeake and Delaware Bay [9]. Norcardiosis, caused by the bacterium *Nocardia crassostreae*, is thought to have originated in Japan and then spread to North America with Pacific oyster transfers [25]. Another renowned and documented case is the introduction of the parasite *Bonamia ostreae* from the United States to Europe. European flat oysters (*Ostrea edulis*) transferred from California to France and Spain appear to be the cause of a devastating crash of the flat oyster population in Europe. In this case, however, the presence of the parasite and the high mortality were known and the seeds transferred were erroneously declared disease-free [29].

The intentional introduction of nonnative bivalves has often resulted in the unintentional transfer of nonnative organisms that "hitchhiked" with the introduced species. Nonnative species can hitchhike within the bivalve, on the bivalve, in water or equipment, in the sediment contained in empty shells or even with other hitchhiking organisms [11]. In San Francisco Bay it has been estimated that 20% of the non-native species resulted from the shipment of eastern (*Crassostrea virginica*) and pacific oyster (*Crassostrea gigas*), many of these species are now important predators and competitors of the resident fauna and flora, as well as pests. In the North Sea it has been estimated that 40% of nonindigenous species hitchhiked with oyster culture, and 43 exotic macroalgae species were introduced in Southern France. This type of introduction contributed historically, at least as much as international shipping to the spread of exotic species [4,9,11,25]. Biofouling organisms have often been introduced with cultured bivalves, especially oysters and mussels. These bivalve species are highly vulnerable to biofouling due to their shells and culture structures thus providing substrate for the settlement of fouling organisms. Pests associated with their transfers include macroalgae (*Codium fragile* spp. *tomentosoides* and *Undaria pinnatifida*), tunicates (*Ciona intestinalis* and *Styela clava*) and gastropods (*Crepidula fornicata*). In some cases, nonnative biofoulers have proliferated, reducing local biodiversity and changing population and community structure in coastal systems. In addition, the bivalves need to be treated with antifouling agents in order to eliminate or reduce the fouling. These treatments are almost universally done over the water. Antifouling agents such as hypochlorite and acetic acid may therefore be added to the environment, and the physical removal of fouling may cause the invasive species to spread or deposit on the bottom [9,11,25].

A particular case of introduction, with repercussion on public health, is the possible transfer of harmful phytoplankton species. It has been proven that the harmful dinoflagellate *Alexandrium* spp. can be transported in the digestive tract and therefore introduced into new environments by mussel, oyster, clam and scallop, especially at the more robust spore and cysts stages. The viability of the algal cells appears to be significantly reduced at 48 hours post-filtration; therefore, a depuration of 48 hours pre-introduction could minimize the risk of transfer of harmful algae [4,11,30].

It is important to point out that the introduction of nonnative bivalve species for aquaculture purposes is now highly regulated by national and international laws greatly reducing the probability of the introduction of diseases and pests. A code of practice for the introduction of non-native species developed by the International Council for the Exploration of the Sea (ICES) has been adopted by many countries and includes preventive measures such as quarantine, extensive disease testing and hatchery breeding so that only first-generation offspring can be released in open water [9,31].

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Effects on marine mammals and seabirds

Bivalve aquaculture operations have the potential to impact marine mammals by causing entanglement from farm structures and litter, changes to prey abundance and partially excluding habitats with disturbance. These impacts have only been identified as potential impacts and not yet demonstrated directly for bivalve culture; the exception being a case of entanglement in mussel spat collectors of two Bryde's whales in New Zealand [9,10,25].

Bivalve culture can affect seabirds due to alteration of food sources, displacement of habitat and noise disturbance [25]. When non-native oyster culture is introduced in soft-sediment areas, an increase in abundance of birds can be noticed, and sea ducks appear to be strongly attracted by bivalve aquaculture operations. During mechanical harvesting of clams an increase in the feeding activities of gulls and waders is observed. In these cases aquaculture created a new habitat for associated fauna [11,27,32]. However, diving ducks are considered one of the most important predators for bivalve culture, particularly mussels, and deliver considerable damage to farmers [10]. They generally predate preferentially on small mussels, however they have been shown to cause damage to both collectors and commercial mussel ropes [33]. It has been calculated that eider ducks can remove up to 2.6 Kg of mussels in a day and the total loss for the farm can be anywhere from 30 up to 75% of the production [10,33]. Many methods have been employed to deter bird predation, including acoustic deterrents, nets, and gunfire [10]. The debate on the most sustainable deterrent is still open. Exclusion nets are currently being studied in order to determine the correct mesh and twine size for different duck species, as shown by Varennes [33].

Evaluation, Control and Prevention of Bivalve Aquaculture Effects

There are different environmental indicators that have been used to evaluate the interaction between bivalves and the surrounding environment. For effective prevention and mitigation the indicators used need to encompass water quality effects, benthic effects, changes in biodiversity, habitat transformation and carrying capacity [34]. There are indicators that help to understand the movement of water and nutrients in the area of interest such as the ones described by Gibbs [14].

- Clearance efficiency: The ratio between number of days that the water takes to clear an inlet and the number of days it would take for the bivalve to process all the water in the inlet.
- Filtration pressure: The ratio between the total carbon extracted by the bivalve in the water column and total carbon fixed by autotrophs.
- Regulation ratio: The phytoplankton turnover rate over the ratio of the daily volume of water cleared by bivalves to the total volume of water.
- Depletion footprint: It can be measured by fluorometer and conductivity-temperature-depth instruments (CTD) [14].

Moreover, nutrient concentration, dissolved oxygen, bacterial abundance, phytoplankton biomass and size can be measured. Other indicators are used to measure the health of sediments and benthic habitat: redox potential, sediment oxygen concentration, similarity indices, biodiversity metrics and indicator species. Finally, there are socio-economic indicators that evaluate the impact and the interest around bivalve culture. For example, social acceptability of the culture, supply availability and livelihood security for the local communities [17,35]. The information obtained from these indicators can be used independently to give a snapshot of the status of the system and the impact of a farm however they should be used to feed a carrying capacity model or a risk-type matrix as part of a larger management plan [14].

Many efforts have been concentrated on developing complex numerical hydrodynamic carrying capacity models, which take into consideration currents, nutrient, plankton and zooplankton [14]. However, most of the modeling focused on production carrying capacity (optimized level of production of the target species), with few on ecological carrying capacity which considers the whole ecosystem. Given the complexity of carrying capacity models and evaluation, they require expertise in many areas and therefore those models are always the result of collaboration between experts in different fields [36]. Modeling is one of the few tools capable of assessing aquaculture sustainability while also considering the cumulative effects of human activities and resident and invasive species. It is therefore the most complete tool to assess and prevent aquaculture impacts on the environment [35]. A good example of modeling for bivalve aquaculture is the complex Farm Aquaculture Resource Management (FARM) model developed by European researchers, which includes information about production and environmental impact for bivalve farms. It combines hydrodynamics, biogeochemistry, population dynamics and economics into a management tool that became a strong decision-support tool for both growers and regulators. This model has been tested for a variety of systems around Europe, from open coast to estuaries, and on a wide range of cultured bivalves such as Pacific oyster, blue and Mediterranean mussel, and Manila clam. The FARM model also shows the indicators of positive impact provided by bivalve culture in helping to reduce eutrophication in the coastal zones [37]. Also, in Canada a bio-physical ecosystem model assessing the environmental effects, particularly regarding nutrient cycling, of bivalve aquaculture in coastal waters has been developed and validated. This model includes both benthic and pelagic components and the cycling of limiting nutrients [38].

Best management practices (BMP) and performance standards have been used as means of prevention for unacceptable environmental interactions and they are often developed by the industry group itself. Regulatory and certification standards can be developed by the public authority and by the buyers. The goal of these standards is a more sustainable, effective and acceptable aquaculture. However, in order to reduce or limit environmental impacts of bivalve culture, they should be developed and implemented at the ecosystem level [9,34].

All the means of prevention and measurement presented above can be gathered into an ecosystem-based management for bivalve culture. Ecosystem-based management is a comprehensive integrated management of human activities implemented to identify and take action on influences that are critical for the ecosystem. It is a tool to achieve sustainable uses of the ecosystem, maintaining its integrity and encompassing interaction between ecological, social and economic systems [35]. In brief, a bivalve culture ecosystem-based management requires models to assess carrying capacity, policies for hazard identification, risk assessment and management, environmental monitoring programs, impact assessment and communication; it should incorporate the best available scientific knowledge, address phytoplankton interactions, impact on the seabed and interactions between farms, consider cost versus benefits, the potential ecological service provided by cultured bivalves, social issues and economic impact [17,35].

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Table 2 summarizes the main environmental effects and their possible means of evaluation and prevention.

Positive effects of bivalve culture

The functions of water clarification and biodeposition that characterize filter-feeding bivalves are valuable providers of ecological services to shallow water ecosystems. Bivalves help buffer estuaries and coastal ocean waters against excessive phytoplankton blooms in response to anthropogenic loading of nitrogen, counteracting the symptoms of eutrophication; they also remove inorganic sediments from suspension, counteracting coastal water turbidity. The biodeposition created by mussels and oysters, through the creation of sediment anoxic microzones where denitrifying bacteria are promoted, induce denitrification, which also help to counteract eutrophication by returning nitrogen into the atmosphere as inert nitrogen gas [3,8,9,39,40]. Moreover, the enhancement of water clarity due to filtration allows deeper light penetration and therefore can increase the growth of seagrasses that are important nursery habitat for many fish, crustaceans and molluscs; bivalves are therefore capable of enhancing estuarine nursery habitats [9,40,41]. These natural functions of bivalves can be employed in aquaculture not only to mitigate the environmental effects of the culture, but also to create added value and services for the surrounding environment.

Restoration

Natural shellfish populations around the world are in decline due to over-exploitation from fisheries, to decline in estuaries condition and in smaller part to diseases introduction [42,43]. As the natural populations decline the important ecosystem services that bivalves provide also drop off and both the water column and the benthic habitat can be affected. It has been therefore suggested that bivalve restoration should be a component of restoring historical baseline conditions and functioning of estuaries. The restoration of oyster in the Chesapeake Bay is the most famous example of bivalve restoration effort [8,9,41,43-45]. Bivalve aquaculture can be considered as an estuarine and coastal ecosystem restoration tool, it could serve to mitigate water quality issues, such as excess chlorophyll and turbidity and even contaminant presence. Although bivalve culture does not provide the same structure created by wild bivalves, culture gears themselves can provide a structural habitat.

It has been suggested that the farmers should receive a compensation for mitigation based on the level of improvement achieved, in addition to selling their product, hence enhancing locally grown seafood production. However, for bivalve aquaculture to work as a restoration tool and give net benefits to the environment, regular removal and responsible disposal of non-native fouling needs to be managed. Moreover, human activities have to be closely controlled to avoid disturbance of any valued species, including birds and marine mammals [9,44].

Re-eutrophication

In Sweden, blue mussels (*Mytilus edulis*) have been used for a study on “agro-aqua recycling”. Cultured mussels were used to reduce the effects of eutrophication created by excess nutrients discharged in coastal waters from farm land runoff and rural living. They were then harvested and re-used as seafood or in agricultural operations as feedstuff and fertilizer [46]. The use of harvested mussels as a substitution of fishmeal in poultry feed was investigated and was successful [46,47]. Moreover, the remainder of mussels and shells were proven to be a valuable land fertilizer, especially interesting for organic farmers who cannot use commercial fertilizers. This model of re-eutrophication, nutrient trading and mussel farming resulted therefore in a successful solution for society, environment and industry [46,48]. The site has been evaluated until 1.5 years after beginning of operation, and in all cases presented a net removal of nitrogen from the system [49]. Research in this field is currently ongoing in other European countries, such as Denmark and the Netherlands.

Integrated multi-trophic aquaculture (IMTA)

Mussels and oysters have been involved in studies regarding polyculture systems. Generally, polyculture or integrated multi-trophic aquaculture (IMTA) combines fed aquaculture such as finfish or shrimp with extractive aquaculture. Extractive aquaculture utilizes filter-feeding organisms, the bivalves, to remove the organic excess nutrients and seaweeds to remove the inorganic excess nutrients, in order to reduce the environmental impact of fed aquaculture. The bivalves perform as biological filters and environmental cleaners. This type of culture is based on the principle that the solution to pollution is not dilution but extraction and conversion [50,51]. A possible further gain can be the fact that it has been experimentally demonstrated that blue mussels (*Mytilus edulis*) ingest sea-lice (*Lepeophtheirus salmonis*) at the copepodid stage and therefore could be a valuable help in controlling sea-lice infestation in farmed salmon [52]. However, Navarrete-Mier [50] showed that excess organic matter produced by a finfish farm in open-water systems was not used by bivalves; the authors conclude therefore that polyculture may not be relevant for diminishing the environmental impact of finfish farms located in areas with high hydrodynamism. It is the opinion of the author that the effect of bivalves in a polyculture system have to be carefully studied in regards to the hydrodynamics of the area during an extended period of time, as the amount of fish waste in the diet of IMTA bivalves varies with season [53]. Critical limitations on the effectiveness of mussels in removing the excess organic nutrients with the current IMTA practices are presented by Cranford [54].

Remediation

Gifford [55] suggested the use of pearl oyster as bio-remediator in polluted environments. The interest in using pearl oysters, such as *Pinctada imbricata* and *Pinctada margaritifera*, for bioremediation is highly attractive since the market value for these species is in the pearl and not in the meat. The authors suggest the possibility to use different pearl oyster species culture in environment polluted by heavy metals, such as lead, copper, zinc and iron; organopollutants, such as PCBs and petroleum hydrocarbons, have been also shown to accumulate in bivalve meat. Finally, pearl oysters can be used as natural filters to clean waters of bacteria, viruses and protozoan from human and animal waste [55]. Pearl oysters have been shown to accumulate significant amounts of pollutants in both meat and shells and therefore they are good candidates for remediation of polluted waters [56]. However, in order to make remediation with bivalve a viable alternative, the disposal of harvested contaminated oysters need to be assessed and addressed. Moreover, tolerance limits of the oyster to certain pollutants need to be investigated, together with their effect on pearl quality [56].

Conclusions

Bivalve aquaculture success is highly dependent on water quality and a healthy ecosystem. There is no doubt that efforts are made to pursue sustainable culture that protects and maintains the supporting environment by operating within the ecological carrying capacity.

The author agrees, however, with Hargreaves [34] when he states that sustainability is not an endpoint, but rather a trajectory of constant improvement. In the author opinion, bivalve culture needs to keep evolving, modifying and striving to couple benefits for the farmers with benefits for the environment. "Constant improvement" also encompasses what research can do for bivalve aquaculture. Polyculture, bivalve restoration and use to counteract human eutrophication are directions worth exploring and ecosystem-based management should be developed and implemented in the areas with bivalve operations. In order to do this, more knowledge regarding the direct effects of bivalve culture on the water column and nutrients is needed; in order to have the most accurate picture of culture impact Nizzoli [13] suggested the necessity to monitor sequentially both suspended culture and benthic environment for nutrients. A constant improvement can be reached only with extensive collaboration between researchers in different fields because the interaction between bivalve culture and the environment is complex and encompasses many disciplines such as biology, ecology, oceanography and social sciences.

Ultimately, as stated by McKindsey [4], there is a need to evaluate which of the effects of bivalve aquaculture on the environment are important and which ones are not, and what we should be managing for. Often negative and positive effects of bivalve aquaculture are strictly related and showed together and often the same effect can be considered both negative and positive, depending on the situation. On balance, whether or not bivalve culture has a negative or positive effect depends on the values that are used to weight the different components [4]. And those values are not a privilege of the industry or the scientists, they have to be chosen in collaboration with the whole society.

Acknowledgement

The author would like to thank Dr. Chris Parrish, Dr. Bill Montevecchi and Dr. Sue Ziegler for the help in developing the topic and for the suggestions to improve this manuscript.

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Ecology, 99(8), 2018, pp. 1802–1814

Oysters and eelgrass: potential partners in a high pCO_2 ocean

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Abstract. Climate change is affecting the health and physiology of marine organisms and altering species interactions. Ocean acidification (OA) threatens calcifying organisms such as the Pacific oyster, *Crassostrea gigas*. In contrast, seagrasses, such as the eelgrass *Zostera marina*, can benefit from the increase in available carbon for photosynthesis found at a lower seawater pH. Seagrasses can remove dissolved inorganic carbon from OA environments, creating local daytime pH refugia. Pacific oysters may improve the health of eelgrass by filtering out pathogens such as *Labyrinthula zosterae* (LZ), which causes eelgrass wasting disease (EWD). We examined how co-culture of eelgrass ramets and juvenile oysters affected the health and growth of eelgrass and the mass of oysters under different pCO_2 exposures. In Phase I, each species was cultured alone or in co-culture at 12°C across ambient, medium, and high pCO_2 conditions, (656, 1,158 and 1,606 μatm pCO_2, respectively). Under high pCO_2, eelgrass grew faster and had less severe EWD (contracted in the field prior to the experiment). Co-culture with oysters also reduced the severity of EWD. While the presence of eelgrass decreased daytime pCO_2, this reduction was not substantial enough to ameliorate the negative impact of high pCO_2 on oyster mass. In Phase II, eelgrass alone or oysters and eelgrass in co-culture were held at 15°C under ambient and high pCO_2 conditions, (488 and 2,013 μatm pCO_2, respectively). Half of the replicates were challenged with cultured LZ. Concentrations of defensive compounds in eelgrass (total phenolics and tannins), were altered by LZ exposure and pCO_2 treatments. Greater pathogen loads and increased EWD severity were detected in LZ exposed eelgrass ramets; EWD severity was reduced at high relative to low pCO_2. Oyster presence did not influence pathogen load or EWD severity; high LZ concentrations in experimental treatments may have masked the effect of this treatment. Collectively, these results indicate that, when exposed to natural concentrations of LZ under high pCO_2 conditions, eelgrass can benefit from co-culture with oysters. Further experimentation is necessary to quantify how oysters may benefit from co-culture with eelgrass, examine these interactions in the field and quantify context-dependency.

Key words: Crassostrea gigas; *disease ecology; ecosystem service; filtration; ocean acidification; plant defense; seagrass.*

INTRODUCTION

Climate change is linked to declining biodiversity, increasing infectious disease, regime shifts, and compromised ecosystem services in both marine and terrestrial systems (Worm et al. 2006, Doney et al. 2009, Altizer et al. 2013, Rocha et al. 2015). The primary driver of climate change is increased atmospheric carbon dioxide (pCO_2). In the ocean, increasing aqueous pCO_2 (coupled with increased atmospheric pCO_2), has caused changes in seawater chemistry leading to a prolonged decrease in ocean and coastal pH, a process referred to as ocean acidification (OA) (Doney et al. 2009, Gledhill et al. 2015). OA results in lower availability of the carbonate ion (CO_3^{2-}) and, therefore, a decrease in the saturation state of calcium carbonate ($CaCO_3$), making it more difficult for marine invertebrates to form calcified shells (reviewed by Hofmann et al. 2010, Kroeker et al. 2010, 2013). Not all organisms are impacted equally by OA (Kroeker et al. 2010, 2013) or by other factors associated with climate change (Howard et al. 2013). Physiological requirements of individual species and altered interactions within ecological communities will determine whether a species is predicted to be a "winner", i.e., experiencing population growth under OA conditions (e.g., seagrasses and algae), or a "loser", i.e., experiencing population declines under OA conditions (e.g., bivalve molluscs, corals, and coralline algae) (Howard et al. 2013, Kroeker et al. 2013, Zimmerman et al. 2017).

In response to observed and predicted effects of OA on sensitive natural and managed populations, numerous agencies are calling for mitigation action plans. For example, in the United States, the Washington State Blue Ribbon

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Manuscript received 6 February 2018; revised 25 April 2018; accepted 3 May 2018. Corresponding Editor: A. Randall Hughes.

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Panel on Ocean Acidification, the West Coast Ocean Acidification and Hypoxia Panel, and the Northeast Coastal Acidification Network are all tasked with summarizing the current science and strategizing plans to mitigate and manage impacts of OA (Gledhill et al. 2015, Chan et al. 2016, WA State Blue Ribbon Panel on Ocean Acidification 2012). Recent studies suggest that, in certain situations, ecosystem level management through biological mitigation of OA may be an effective strategy for increasing resilience of coastal ecosystems to OA (Billé et al. 2013). At local scales, the co-cultivation of seagrass or macro-algal populations with OA-sensitive taxa is hypothesized to improve the health of the sensitive taxa (Unsworth et al. 2012, Koch et al. 2013, Hendriks et al. 2014). This process, called phytoremediation, results when dissolved inorganic carbon is removed from the water column during photosynthesis, resulting in an increase in seawater pH (Zimmerman et al. 2007, Koch et al. 2013, Hendriks et al. 2014). The photosynthetic activity of seagrasses is predicted to create an OA buffer that extends beyond the seagrass canopy (Hendriks et al. 2014), potentially providing refuge for pH-sensitive organisms that reside within and close to these meadows. More research is required to understand when phytoremediation of OA by seagrasses is effective.

The hypothesized role of seagrasses in mitigating coastal OA may be compromised by the numerous threats that are leading to global seagrass declines (Orth et al. 2006). In some temperate zones, the dominant seagrass species, eelgrass (*Zostera marina*), is threatened by eelgrass wasting disease (EWD), caused by pathogenic strains of the opportunistic protist, *Labyrinthula zosterae* (LZ; Muehlstein et al. 1991, Sullivan et al. 2013, Martin et al. 2016). LZ infection causes the rapid development of necrotic lesions on eelgrass blades (EWD), which can lead to mortality in severe cases (e.g., Groner et al. 2014). EWD has been associated with rapid declines of eelgrass populations in the past, including an estimated 90% reduction of eelgrass populations in the north Atlantic in the 1930s (Renn 1935). EWD outbreaks in the northwest Atlantic in the 1980s were associated with eelgrass population declines (Short et al. 1987). Currently, in the Northeast Pacific, infectious strains of LZ have been isolated from eelgrass blades exhibiting signs of EWD, indicating a potential role in local eelgrass declines (Groner et al. 2014, 2016, Martin et al. 2016). The influence of increased pCO_2 on the resilience of eelgrass to EWD or infection by LZ is unknown, although OA is a growing issue in this area (Feely et al. 2010). While higher pCO_2 is frequently associated with increased photosynthesis (Koch et al. 2013), the production of phenolic compounds, which are hypothesized to be key defenses against EWD, have been shown to decrease in eelgrass at higher pCO_2 for, as yet, undetermined reasons (Buschbaum et al. 1990, Vergeer and Develi 1997, Arnold et al. 2012). It is unclear how EWD is affected by OA conditions; EWD could compromise benefits that eelgrass is hypothesized to provide to OA-sensitive taxa.

Recent studies suggest that filter-feeding molluscs can reduce pathogen concentrations in the water column (e.g., Faust et al. 2009, Webb et al. 2013, reviewed in Burge et al. 2016), leading to the intriguing hypothesis that filter-feeders may be able to reduce transmission of EWD. It is unknown whether filter-feeding molluscs may reduce concentrations of LZ in the water column, or if this process may be compromised at high pCO_2. Many species of oysters, including Pacific oysters, frequently co-occur with eelgrass, either in naturalized populations or in aquaculture (Wall et al. 2008, Wagner et al. 2012) making them an obvious candidate for investigating this hypothesis.

The goal of this research was to examine how culturing eelgrass and Pacific oysters together affects the health and growth of each species under a variety of pCO_2 conditions (Fig. 1). We used a two-phase experiment conducted in controlled laboratory conditions. In Phase I, we exposed separate and mixed cultures of eelgrass and oysters (here-after referred to as ‘cultured alone’ or ‘co-cultured’) to a range of pCO_2 conditions and tested the hypotheses that (Fig. 1):

1. Eelgrass growth will be greater at high pCO_2 than at ambient pCO_2
2. Eelgrass will have lower EWD severity when co-cultured with oysters
3. Increased pCO_2 will alter the severity of EWD (contracted in the field prior to experiment)
4. Eelgrass will increase daytime pH
5. Oyster mass will be lower in the absence of eelgrass and at higher levels of pCO_2

In Phase II of the experiment, we conducted a challenge experiment, exposing eelgrass from phase I to LZ or a control. We tested the hypotheses that (Fig. 1):

6. Eelgrass co-cultured with oysters will have lower EWD severity and pathogen LZ loads than eelgrass cultured alone
7. EWD severity and LZ loads will be higher at high pCO_2
8. Production of phenolic compounds and condensed tannins will be lower in diseased eelgrass and at high pCO_2.

Methods

Experimental trials

Experiments were conducted in the Ocean Acidification Experimental Laboratory (OAEL) at the University of Washington’s Friday Harbor Laboratories (FHL) (48.550° N, 123.008° W), University of Washington, on San Juan Island, Washington state.

Specimens

Vegetative (i.e., non-reproductive) eelgrass shoots were collected in the intertidal zone at Beach Haven, Orcas Island, WA, USA (48.691° N, 122.952° W) on 21 August 2014. This site was chosen because it has a relatively low EWD prevalence for this region (mean ± SE: 19 ± 3% diseased on 9 August 2014; Groner et al. 2016) and a stable eelgrass population has existed since at least 2010 (Wyllie-Echeverria et al. 2010, Groner et al. 2014, 2016). After collection, the shoots were kept in cool seawater until returning to FHL where they were stored in continuously flowing seawater (~30 ppt, ~12°C) and rhizomes were clipped to separate shoots into unique ramets containing a meristem with associated leaves and several cm of rhizome. Grazers, egg

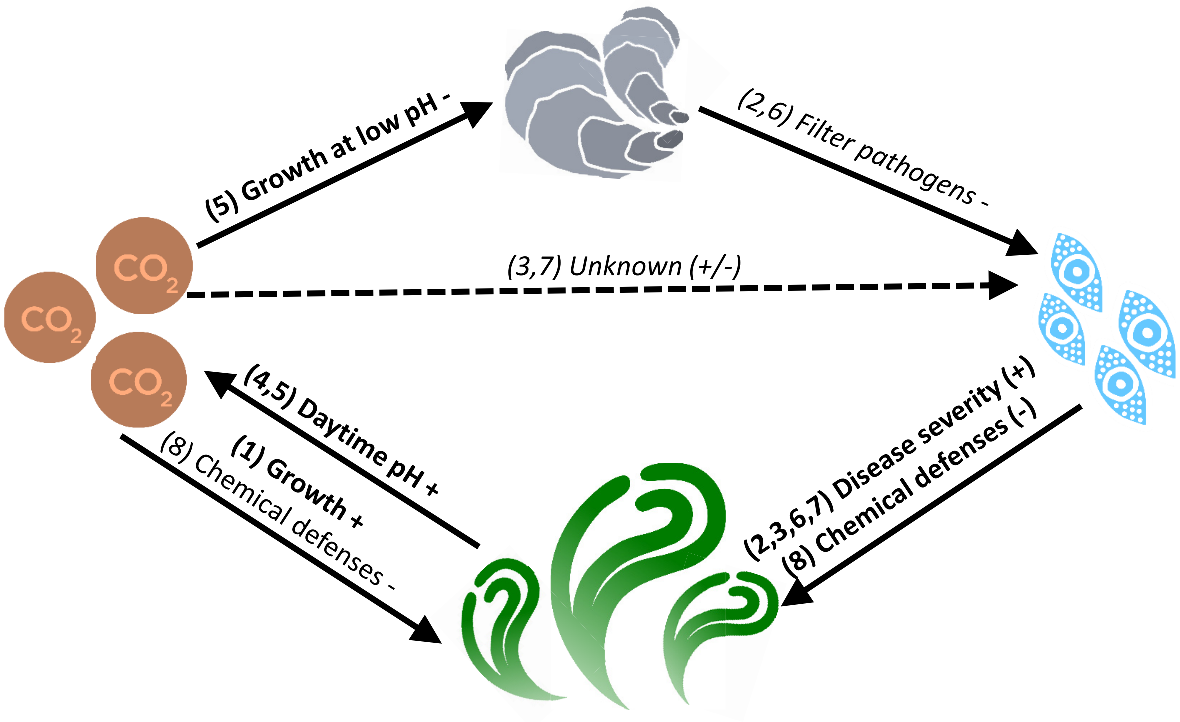

FIG. 1. Conceptual diagram showing some of the hypothesized effects of increased pCO_2, exposure to pathogens and co-culture on two co-occurring ecosystem engineers, Pacific oysters and eelgrass, on each other. Effects are labelled according to the hypotheses listed numerically in the introduction along with the anticipated effect (+ or −). Note that some hypotheses involve indirect effects mediated through two arrows (i.e., 2, 3, 5, 6 and 7). Direct effects on *Labyrinthula* spp. (italicized) were not measured independently of eelgrass in this study.

masses and diseased or senescing leaves were carefully removed from these shoots.

Diploid juvenile *C. gigas* (~4 months of age and 25–30 mm in length) were provided by the Taylor Shellfish Farms from their floating upwelling system in Shelton, WA. These oysters came from single stock and were raised in buffered seawater, so they had not previously been exposed to OA conditions.

Juvenile oysters were transported on ice to FHL, and were maintained in aerated, flowing seawater for two weeks at ambient temperatures (~12°C) and fed as described below.

Specimen maintenance during experiments

For the experiments (described below) oysters and eelgrass were held in 3.5 L chambers housed within 95 L coolers to maintain temperature and each received aerated, flowing seawater (~28–30 ppt, 12°C, 70 min seawater residence time). Oysters were batch-fed daily with Shellfish Diet 1800® (Reed Mariculture) at a total estimated concentration of 2×10^2 cells/mL. Full-spectrum LED lights for aquatic plants (MarineLand, Blacksburg, VA) were placed above each cooler to provide the tanks with 161 ± 3 (mean ± SE) $\mu mol\ photons \cdot m^{-2} \cdot s^{-1}$ below the water surface. Lights were set on a 14:10 light:dark schedule. To reduce algal blooms, which could clog plumbing and block light, tanks were treated daily with 0.67 ppm (final concentration) of Germanium dioxide (GeO_2; Markham and Hagmeier 1982). This concentration of GeO_2 kept diatom growth within densities found in nature; without herbivores in our system, overgrowth would have occurred. GeO_2 inhibits diatom growth by preventing uptake of silica. No deleterious effects of GeO_2 on focal organisms were expected at these low concentrations. Replicates were rotated within each cooler once a week to control for any potential spatial effects.

Experimental set-up

Phase I trials.—The goal of phase I experiments was to compare the influence of varying pCO_2 on two taxa individually (eelgrass only and oysters only) relative to co-culture of both species. This phase consisted of three biotic treatments, referred to as ‘‘oyster”, ‘‘eelgrass” and ‘‘co-culture” treatments, crossed with three pCO_2 treatments, referred to as ambient, medium or high levels (Fig. 2a). Each of the nine treatments was replicated five times for a total of 45 experimental units. Each oyster replicate contained 32 juvenile oysters (25–30 mm in length). Eelgrass treatments contained 23–24 g of eelgrass ramets (wet weight of leaves, rhizomes and roots, 4–6 ramets/replicate) that were zip-tied together with a small glass weight at the rhizome in order to keep the ramets oriented vertically. Co-culture treatments contained both juvenile oysters ($n = 32$) and eelgrass (23–24 g). Experimental densities of eelgrass and oysters are within the range found in the Salish Sea (*personal observation*).

The pCO_2 treatments were designed to mimic current pCO_2 conditions in the San Juan Islands, Washington (656 ± 199 μatm = ambient treatment, mean ± standard deviation [SD]) and two future scenarios (1,158 ± 73 and

FIG. 2. Experimental design for phase I (A) and phase II (B) of our experiment. pCO_2 was manipulated at the level of the cooler (bigger rectangles), while biological treatments (smaller squares) were replicated within coolers. Thus, in phase I, there were a total of 45 experimental units (smaller squares) which represented nine experimental treatments replicated five times each. In phase II, there were a total of 32 experimental units (smaller squares), representing eight experimental treatments replicated four times each. For all analyses, measurements were averaged within the level of the experimental unit (small squares) and a random effect for the cooler was included in order to account for the non-independence of replicates within the coolers.

TABLE 1. Water quality conditions for phase I and phase II experiments.

pCO_2 Treatment	Salinity (ppt)	Temperature (°C)	Total alkalinity (μmol/kgSW)	pH	pCO_2 (μatm)	CO_3^{2-}	Ω_{CAL}	Ω_{ARAG}
Phase I								
Ambient	30.4 ± 0.2	12.2 ± 0.7	2104.4 ± 7.1	7.84 ± 0.12	655.5 ± 198.8	81.4 ± 20.5	1.99 ± 0.50	1.26 ± 0.32
Medium	30.4 ± 0.2	11.9 ± 0.5	2102.5 ± 4.4	7.60 ± 0.03	1157.5 ± 72.8	46.8 ± 3.8	1.15 ± 0.09	0.72 ± 0.06
High	30.4 ± 0.3	11.7 ± 0.4	2102.2 ± 4.4	7.26 ± 0.01	1605.9 ± 107.8	21.9 ± 0.8	0.53 ± 0.19	0.34 ± 0.01
Phase II								
Ambient	30.6	15.1 ± 0.1	2,116	7.89	488.2	110.2	2.71	1.72
High	30.6	15.2 ± 0.2	2,109	7.30	2,013	33.6	0.83	0.53

Notes: Means ± SD are presented. With the exception of temperature, water quality parameters were only taken once during phase II, so measures of variance were not possible.

1,606 ± 108 μatm = medium and high treatments, respectively; Feely et al. 2010). These treatments corresponded to pH levels of 7.84 ± 0.12, 7.6 ± 0.03 and 7.26 ± 0.01 (mean ± SD, Table 1). The pCO_2 levels in the medium and high treatments are higher than the end-of-century projections for the 'rapid economic growth' (IPCC 'A1' scenarios, IPCC 2007, 2014). However, within the San Juan Archipelago, seawater pCO_2 is consistently higher and more variable than global averages. For example, pCO_2 averaging 700 (pH = 7.8), but ranging from lower than 400 to higher than 1,000, have been observed in seawater collected weekly between July 2011 and July 2013 from coastal waters within 20 seaway km of our eelgrass collection site and <1 km from other eelgrass beds (Murray et al. 2015). These data suggest that these '"future" scenarios will occur soon in the San Juan Archipelago. The pCO_2 treatments were nested within six coolers. Each cooler had a single source of pCO_2 treated water and contained up to eight experimental units. Thus, duplicate coolers for each pCO_2 treatment held either two or three replicates of each culture treatment (for a total of five replicates per treatment (Fig. 2a). The temperature was set to 12°C in each cooler. Total alkalinity (a conserved thermodynamic measure of seawater acid-base chemistry that is equal to the charge difference between conservative cations and anions, Doney et al. 2009) and saturation states for aragonite and calcium carbonate are shown in Table 1. See

Appendix S1 for methods used to maintain pCO_2 levels and quantify carbonate chemistry.

Phase I of the experiment began on 28 August 2014 (experiment day 1) and continued until 18 September 2014. To understand how eelgrass affected pH, we measured the daytime pH and TA of each replicate every 3–4 d. On experiment day 20 we measured pH every 3 h over a 24-h period beginning at 12 pm in order to understand how light availability influenced pH measurements. On the last day of the experiment, we measured eelgrass mass, growth and health, and oyster mass and length. Eelgrass mass measurements consisted of the combined mass of eelgrass ramets within each replicate. In order to measure linear eelgrass growth, we put a pinprick in each shoot at the base of the rhizome (on 11 September) and then measured the distance on the youngest leaf from the pinprick to the rhizome base a week later (on 18 September) (Dennison 1990). For eelgrass health, we assessed disease status and severity of each eelgrass shoot by counting the number of leaves with lesions and dividing this by the total number of leaves (EWD shoot severity, see Appendix S1 for more details). Any EWD lesions that developed on the collected eelgrass were from asymptomatic infections that occurred in these shoots prior to collection. No source of LZ-free plants was available. The mass of each oyster was measured after drying the shell. Our experimental design included random placement of plants in tanks and random assignment of tanks to treatment so that infections in vegetative leaves would be randomly distributed across replicates.

Phase II.—On 18 September, after 24 d of acclimation to pCO_2 conditions (during phase I), we conducted a 10-d pathogen challenge. From the ambient and high pCO_2 treatments, the four healthiest eelgrass ramets from each of four replicates in the eelgrass and co-culture treatments were selected for the disease trial (all eelgrass ramets remained in ambient and high treatments, respectively). Eelgrass ramets with lesions were pruned, until only visually healthy ramets remained. Each of these 16 experimental units was then divided into two, with each resulting replicate containing two eelgrass ramets, and, in the co-culture treatments, 15 oysters. One set of 16 replicates was exposed to a 6×10^4 cells/mL of LZ (isolate 8.16.D) in 0.22 μm filtered sea water (FSW) and the other set of 16 replicates was exposed 0.22 μm FSW as a sham challenge (negative control; Fig. 2b). Previous research revealed that, in most cases, this concentration of cells is required to produce an infection experimentally (Groner unpublished data). Isolate 8.16.D was isolated in 2011 from an EWD lesion on eelgrass that originated from Picnic Cove, Shaw Island, WA (48.565° N, 122.924° W). The strain was cultured on serum seawater agar (Muehlstein et al. 1991), and has been shown to be infectious in prior experiments (Groner et al. 2014). Prior to inoculation, LZ cells were harvested, de-clumped by vortexing and adjusted to a final concentration of 6×10^4 cells/mL.

Eelgrass ramets were challenged in static (no water flow), aerated, containers with 200 mL seawater for 24 h at 15°C. All replicates were dosed with 33 units/mL Penicillin and 33 ug/mL Streptomycin at the time of the disease challenge in order to prevent bacterial coinfection (e.g., Burge and Friedman 2012). After 24 h, the eelgrass ramets and oysters were moved to the 3.5 L experimental units with flowing seawater (as described above). All treatments were held at 15°C. We used this temperature to represent summer temperatures (when EWD prevalence is highest in the San Juan Archipelago, WA, USA; Groner et al. 2014, 2016). We attempted to maintain similar pH values to those found in phase I (which was run at 12°C as compared to 15°C in phase II), which resulted in pCO_2 inputs of 488 and 2,013 μatm to the ambient and high treatments and corresponded to pH levels of 7.89 and 7.30, respectively. While each phase had different pCO_2 inputs, the ambient treatments in phases I and II are within the range of values currently being recorded in coastal waters in the San Juan Archipelago (Murray et al. 2015). The difference in values between treatments reflects the lower solubility of pCO_2 at higher temperatures and greater (per unit) influence on pH.

The experiment was terminated on 27 September 2015. At this point, all eelgrass leaves were photographed for a visual assessment of disease status. Samples from each eelgrass ramet were taken for quantification of pathogen load using quantitative PCR (qPCR) and measurement of defenses. To further quantify EWD, we calculated the area of all lesions on the longest leaf of each plant relative to the total leaf area (LL severity). qPCR samples were taken from the longest leaf of each shoot. If the longest leaf was diseased, the DNA sample was collected with at least part of one lesion. Samples that were selected for qPCR were stored in 70% molecular grade ethanol. The second longest leaf was frozen at −20°C and moved to −80°C within the day for measurements of total phenolic and condensed tannin concentrations.

See Appendix S1 for general methods on the calculation of carbon chemistry, visual assessment of EWD, quantitative PCR of LZ, and measurements of total phenolic compounds and condensed tannins.

Statistical analysis

Phase I.—We used linear mixed effect models (package 'lme4' in R v. 3.3.1) to test each of hypotheses 1–5. We tested the independent and interactive effects of pCO_2 and culture treatments on replicate (i.e., tub) mean values of eelgrass growth, severity of EWD and oyster mass. For each model, all combinations of fixed effects (including interactions and null models) were run. All models were fit with maximum likelihood and included a random effect for water source (i.e., cooler) such that:

$$\text{Response} = \mu + \beta_1 \text{LZ} \times \beta_2 \text{pCO}_2 \times \beta_3 \text{Culture} + \mu_{\text{water source}} + \varepsilon_{ij}$$

where β indicates effect sizes for fixed effects, μ indicates the grand mean and random intercepts for water source and phase I treatments, and ε indicates an error term.

For each analysis, we used model selection based on minimizing Akaike Information Criteria for finite samples (AICc) values to quantify the separate model fits (package 'MuMIn'). The models with the lowest values were considered to have the most support. However, when the ΔAICc (i.e., the difference in AICc values for the model being evaluated and the best-fitting model) was <3, the models were considered to have similar statistical support (Burnham and Anderson 2002). In those cases, we presented the formulas for all supported models along with their AICc values and weights. We then calculated and interpreted the results of the weighted conditional average of all models with a ΔAICc less than three (Burnham and Anderson 2002). When multiple models are supported (i.e., ΔAICc for several models is <3), high model selection uncertainty exists and selection of a single 'best' model can result in the loss of information. Calculation of a weighted average model is a more conservative approach that allows information from all competing models to be incorporated (Burnham and Anderson 2002). Note that when models are averaged, significance tests are based on *z*-values, however if there is only a single best model, significance tests are based on *t*-values.

Phase II.—We tested hypotheses 6–8 using linear mixed effects models (described above). We tested the independent and combined effects of oyster presence, pCO_2 and LZ on replicate means of disease severity, pathogen quantity (from qPCR), total phenolic compound concentrations and condensed tannin concentrations. Pathogen quantity was log10 transformed prior to analysis. All models included a random effect of the water source and an additional random effect for the replicate in Phase I from which samples originated. Thus:

$$\text{Response} = \mu + \beta_1\text{LZ} \times \beta_2\text{pCO}_2 \times \beta_3\text{Oyster} + \mu_{\text{water source}} + \mu_{\text{phase I tub}} + \varepsilon_{ij}$$

Model selection was performed as described for phase I.

For all statistical tests, we graphed the predicted (model) results as opposed to the original data, in order to isolate the treatment effects from possible block effects due to differences in coolers. Original data are graphed in Appendix S1 (Figs. S1–S4).

Data and statistical code are available on figshare at https://doi.org/10.6084/m9.figshare.6182522 (Groner et al. 2018).

RESULTS

Phase I

Leaf elongation was greatest in the medium pCO_2 treatment (Fig. 3, Appendix S1: Fig. S1). The best-fit models for leaf elongation included terms for pCO_2, oysters and the null model (Table 2a). The weighted average model showed the leaf elongation was 11 mm greater at medium pCO_2 than at ambient pCO_2, and 6 mm greater at high pCO_2 than at ambient pCO_2; differences in leaf elongation were only significant at medium pCO_2. Oyster presence was included in the average model, however, it did not significantly influence leaf elongation.

At the end of phase I, the prevalence of EWD was 68%, 75% and 51% in eelgrass in the absence of oysters at ambient, medium and high pCO_2, respectively and 64%, 49% and 32% in the presence of oysters at ambient, medium and high pCO_2, respectively. Due to the low sample size of eelgrass plants, we only ran statistics on EWD severity and not prevalence.

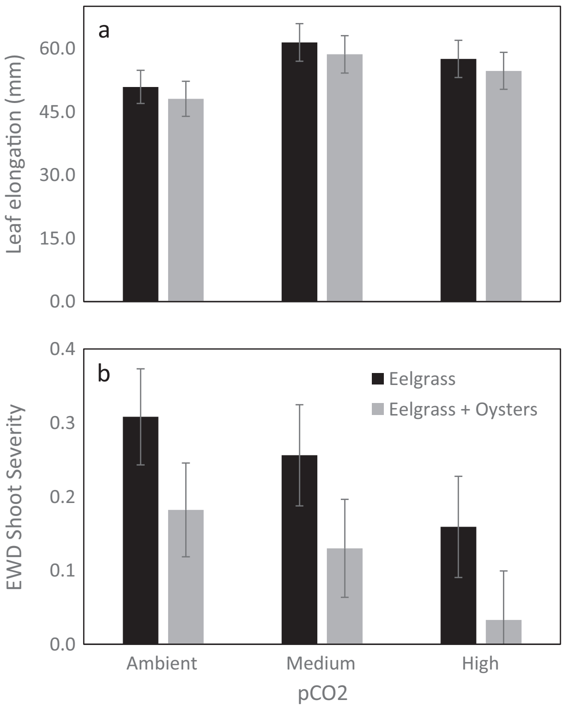

FIG. 3. Phase I: Predicted effects of pH and oyster presence on leaf elongation (a) and the proportion of leaves with lesions (b) after 3.5 weeks. Predictions shown control for block (i.e., cooler) effects. Means ± standard errors are shown.

The EWD shoot severity was lower in the presence of oysters and at high pCO_2. The best-fit models for EWD shoot severity included terms for oysters alone, the null model and oysters + pCO_2 (Table 2b, Fig. 3). The EWD shoot severity in the eelgrass monoculture at ambient pCO_2 was 31 ± 7%. The weighted average model showed that EWD shoot severity was in the presence of oysters than in the absence of oysters (by 13 ± 7%, $P < 0.05$) and at high pCO_2 relative to ambient pCO_2 (by 15 ± 7%, $P < 0.05$).

Daytime pH in the treatments was influenced by the pCO_2 of the input water and the co-culture treatments. The best-fit model for the effect of treatments on the change in daytime pH included terms for both the pCO_2 of the input water and co-culture treatments, but not their interaction (Figs. 4, 5, Appendix S1: Fig. S2). The co-culture treatment caused seawater pH in the treatment to increase in the daytime by 0.05 ± 0.01 pH units relative to that of the inflowing seawater ($t_{8.7} = 3.8$, $P = 0.004$). Eelgrass presence alone caused seawater pH to increase in the daytime by 0.09 ± 0.01 units ($t_{39.8} = 4.4$, $P < 0.0001$), while oyster presence alone caused a decrease in daytime seawater pH by 0.03 ± 0.01 pH units relative to that of the incoming seawater ($t_{39.9} = -8.6$, $P < 0.0001$). The magnitude of change in daytime pH was significantly greater in the medium and high pCO_2 treatments than the ambient pCO_2 treatment (by 0.11 ± 0.02 and 0.18 ± 0.02 pH units, respectively, $t_7 = 7.3$ and $t_{6.8} = 11.9$, respectively, both $P < 0.001$). No other models for daytime pH had a ΔAICc < 3 from the best model.

Final oyster mass was lower at high pCO_2. The best-fit models for oyster mass included pCO_2 alone (AICc = −74.1, weight = 0.364), the null model (AICc = −73.8,

TABLE 2. Phase I results: Best models (ΔAICc < 3) for shoot severity of eelgrass wasting disease (EWD; percent of leaves with lesions) and leaf elongation are shown, followed by the weighted average of the top models.

	AICc	weight	Estimate	Adjusted SE	*z*-value	*P*-value	Model
(A) Leaf elongation							
Model							
Null	229.5	0.516					
pCO$_2$	230.5	0.316					
Oysters	231.8	0.168					
Weighted average model							
Intercept			**50.88**	**3.94**	**12.9**	**<0.0001**	
Oysters			−2.83	4.35	0.6	0.52	
Medium pCO$_2$			**10.54**	**5.01**	**2.1**	**0.04**	
High pCO$_2$			6.63	4.88	1.4	0.17	
(B) Eelgrass wasting disease (EWD): shoot severity							
Model							
Oysters	−13.3	0.508					
Null	−11.9	0.252					
Oysters + pCO$_2$	−11.8	0.24					
Weighted average model							
Intercept			**0.308**	**0.065**	**4.7**		**<0.0001**
Oysters			**−0.126**	**0.062**	**2**		**0.04**
Medium pCO$_2$			−0.052	0.072	0.7		0.47
High pCO$_2$			**−0.149**	**0.072**	**2.1**		**0.04**

Notes: Model weights are given next to AICc values. Significant effects ($P < 0.05$) are in bold.

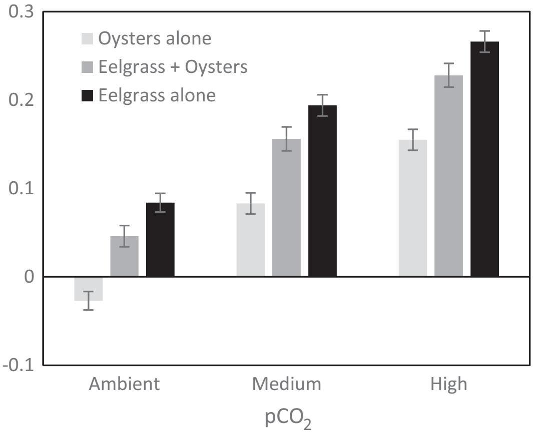

FIG. 4. Phase I: Predicted change in daytime pH across the treatments after controlling for block effects. Means ± standard errors are shown.

weight = 0.323), eelgrass alone (AICc = −72.5, weight = 0.166) and pCO$_2$ + eelgrass (AICc = −72.3, weight = 0.148), (Fig. 6, Appendix S1: Fig. S3). In the averaged model, oyster mass was significantly lower (by 74 ± 27 mg, $z = 2.8$, $P < 0.01$) in the high pCO$_2$ treatment as compared to the ambient conditions (which was 1940 ± 30 mg). Oyster mass tended to be lower in the medium pCO$_2$ treatment as compared to ambient conditions (by 46 ± 27 mg), however this was not significant ($z = 1.7$, $P = 0.08$). Co-culture of oysters with eelgrass did not significantly alter their mass at the end of the experiment ($z = 1.1$, $P = 0.26$).

Phase II

At the end of phase II, LZ exposure increased pathogen load, both LZ and the LZ-by-pCO$_2$ interaction altered EWD shoot severity (Figs. 7a-b, Appendix S1: Fig. S4a,b). The best-fit models for EWD shoot severity at the end of phase II included terms for the LZ-by-pCO$_2$ interaction, LZ + pCO$_2$, oysters + LZ + pCO$_2$, LZ alone and oysters + LZ (Table 3). The weighted average model showed that at ambient pCO$_2$, LZ exposure significantly increased EWD shoot severity from 25 ± 8% to 65 ± 8%. At high pCO$_2$, the effect of LZ was much smaller, only increasing EWD shoot severity from 15 ± 8% to 36 ± 8%. No other terms in the model were significant.

Some control plants in phase II (not experimentally exposed to LZ) developed low levels of EWD. In the control treatments, LL severity was 1.0 ± 0.5% and 1.5 ± 0.6% in the ambient and high pCO$_2$ treatments, respectively. In contrast, in the LZ exposed treatments, LL severity 7.7 ± 3% and 3.3 ± 1.5% in the ambient and high pCO$_2$ treatments, respectively. Lesion morphology did not visibly differ across treatments.

Exposure to LZ increased the intensity of LZ infection at the end of Phase II. The best-fit models for the log10-transformed qPCR results included terms for LZ + pCO$_2$, the LZ-by-pCO$_2$ interaction, and LZ alone (Table 3b, Fig. 7b). In the weighted average model, LZ presence significantly increased the pathogen load from an average of 2 to 20 LZ cell equivalents per mg of dry eelgrass tissue ($P < 0.0001$). While, only a trend ($P = 0.10$), the impact of LZ exposure on pathogen load was nearly an order of magnitude lower at high pCO$_2$ than at ambient pCO$_2$.

The concentration of condensed phenolic compounds in the eelgrass ramets at the end of phase II was significantly higher in the presence of LZ relative to the controls and significantly lower at high pCO$_2$ relative to ambient pCO$_2$ (Table 4b, Figs. 7c., Appendix S1: Fig. S4c). LZ exposure increased the concentration of phenolic compounds in eelgrass ramets (% by dry mass) by 24 ± 9% relative to eelgrass ramets not exposed to LZ. High pCO$_2$ decreased the

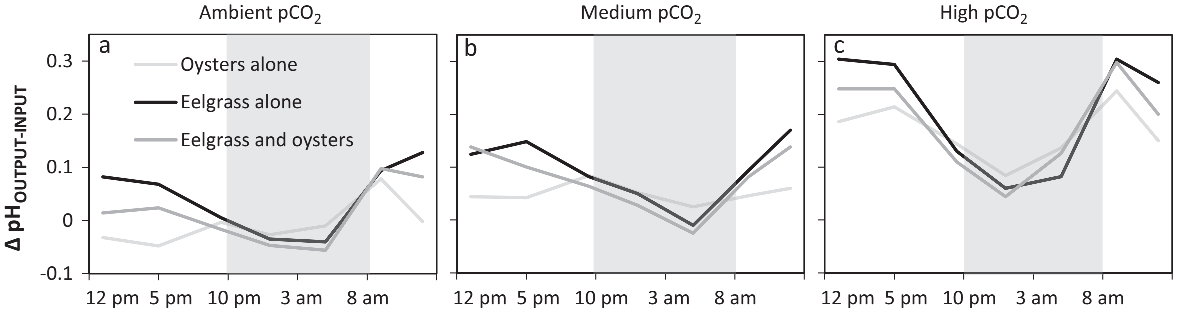

FIG. 5. Phase I: Average change in pH in the inflow and outflow of experimental units containing eelgrass only, oysters only and eelgrass and oysters together across a 24 h period. The light: dark schedule is indicated with the shading.

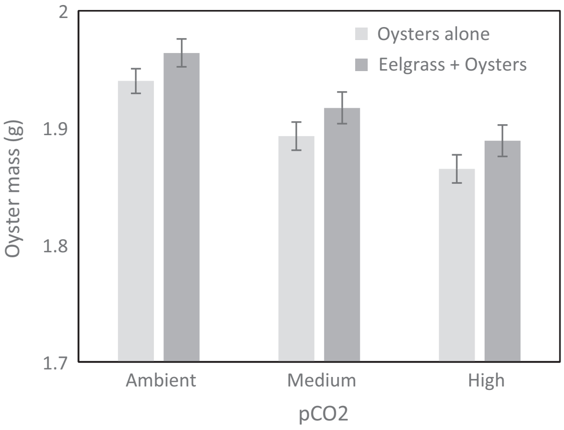

FIG. 6. Phase I: Predicted oyster mass after 3.5 weeks of exposure to different pCO_2 treatments in the presence or absence of eelgrass after controlling for block effects. Means ± standard errors are shown.

phenolic concentration in the eelgrass ramets by 25 ± 10% relative to the phenolic concentration found in eelgrass ramets exposed to ambient pCO_2. While the weighted average model included the LZ × pCO_2 interaction, it was not significant. The best fit models for the % eelgrass dry mass consisting of phenolic compounds included the LZ + pCO_2, LZ alone, and the LZ-by-pCO_2 interaction.

The concentration of condensed tannins in the eelgrass ramets was lower in the high pCO_2, LZ exposure treatment. The sole best-fit model identified for tannin concentrations (Table 4, Fig. 7d, Appendix S1: Fig. S4d) included a significant LZ-by-pCO_2 interaction such that, relative to all other treatments, the concentration of condensed tannins (% by dry mass) was 6.7 ± 2% lower for eelgrass ramets exposed to LZ and high pCO_2. None of the other terms in the model were significant.

DISCUSSION

Our study demonstrated that increased pCO_2 in seawater can alter growth rates of native (eelgrass) and non-native (Pacific oysters) ecosystem engineers and influence host-parasite interactions. In the first phase of our experiment, both increased pCO_2 and co-culture with oysters benefitted eelgrass. Eelgrass experienced less severe lesions from natural LZ infections under high pCO_2 and in the presence of oysters. The mechanism for the effect of pCO_2 on EWD severity is not known. We hypothesize that the effect of oysters was a result of filtering infective LZ out of the water column. The presence of eelgrass increased daytime seawater pH; however, this effect was not substantial enough to counteract the negative effects of high pCO_2 on oysters as measured by oyster mass. Oyster mass was lower in both the presence and absence of eelgrass at high pCO_2 relative to ambient pCO_2. In the second phase of our experiment, when we conducted a pathogen challenge using high concentrations of LZ, eelgrass ramets exposed to both high pCO_2 and LZ had decreased condensed tannins. High pCO_2 reduced total phenols while LZ exposure increased total phenols. Further research is necessary to understand whether these changes represent an adaptive response to infection. Consistent with phase I results, LZ exposure increased EWD severity and this effect was reduced at high pCO_2. In contrast with phase I, oyster presence did not have an effect on EWD severity, possibly because the high experimental dose of LZ swamped any potential effect of filtration. Overall, these results demonstrate the importance of multi-species interactions in modulating the effects of changing seawater chemistry on ecosystem engineers (Kroeker et al. 2013).

The reduction in severity of EWD at higher pCO_2 in both phases of the experiment suggests that the eelgrass-LZ interaction is sensitive to carbonate chemistry. It is unclear whether the host, the pathogen or both are driving this shift. While eelgrass experienced higher growth in the medium and high pCO_2 treatments (measured in phase I), defense-related compounds (phenolics and condensed tannins; both measured in phase II) were reduced at high pCO_2 alone (phenolics) or in combination with LZ exposure (condensed tannins) relative to those in ambient conditions. Although all of these parameters were not measured in both phases of our study, observed trends indicate possible trade-offs between defenses and growth and should be further investigated. The ranges of EWD severity observed in phase I (natural infection) are typical of those observed at sites in the San Juan Archipelago during the summer (which our experimental conditions mimicked, Groner et al. 2014, 2016). The effect of OA on seagrass diseases likely varies with the pathogen. For example, it has been hypothesized that some pathogens of seagrasses including the oomycete *Halophytophthora* sp. may be expected to increase in severity with OA (Sullivan et al. 2017). However, as with most infectious marine diseases, the effects of OA on infectious diseases of seagrasses are not well explored. In contrast to our results, OA has been found to increase infectious disease in other

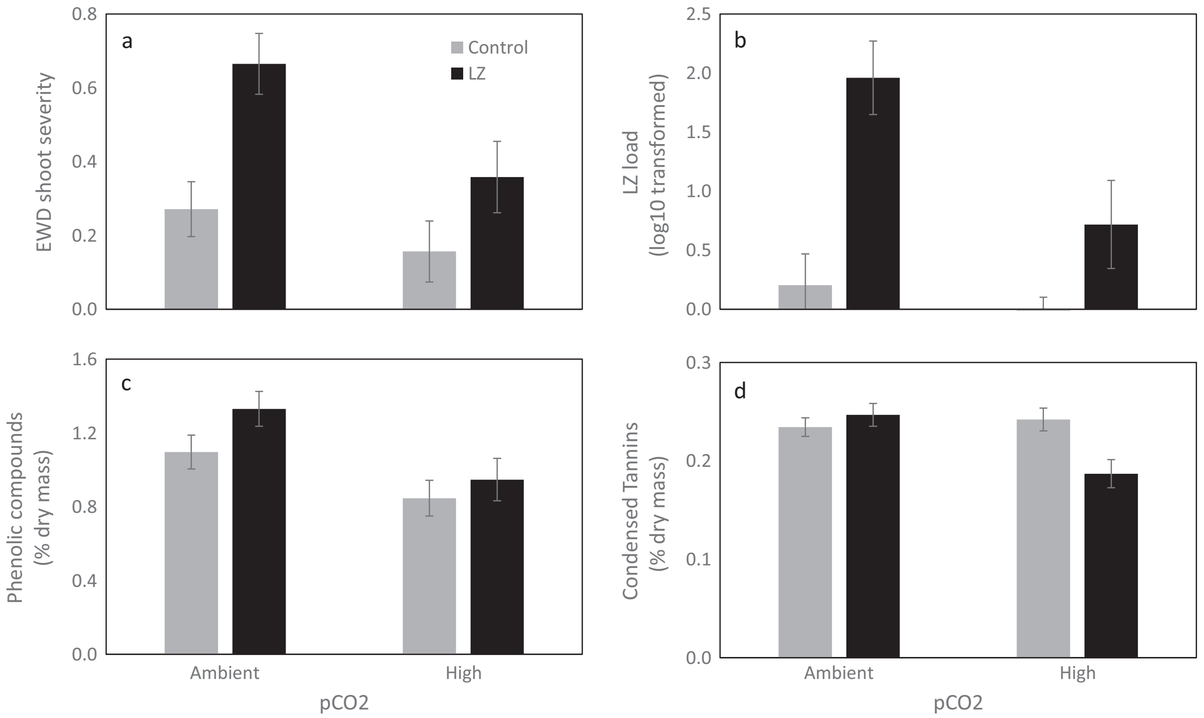

FIG. 7. Phase II: Predicted effects of elevated pCO$_2$ and exposure to LZ on the proportion of eelgrass leaves with lesions (a), pathogen load (b), concentration of phenolic compounds (c) and concentration of condensed tannins (d) after controlling for block effects. Means ± standard errors are shown. No statistical differences were found between treatments with and without oysters. These treatments are pooled in the figure.

TABLE 3. Phase II results.

	AICc	Weight	Estimate	Adjusted SE	*z*-value	*P*-value
(A) Eelgrass wasting disease: shoot severity						
Model						
LZ $\times$ pCO$_2$	−4.7	0.29				
LZ + pCO$_2$	−4.3	0.24				
LZ	−3.7	0.18				
Oysters + LZ $\times$ pCO$_2$	−2.6	0.11				
LZ	−2.5	0.10				
Oysters + LZ	−2.0	0.08				
Weighted average model						
Intercept			**0.30**	**0.07**	**4.1**	**<0.0001**
LZ			**0.40**	**0.08**	**5.1**	**<0.0001**
High pCO$_2$			−0.10	0.09	1.0	0.30
LZ $\times$ high pCO$_2$			**−0.19**	**0.09**	**1.9**	**0.05**
Oysters			−0.09	0.08	1.2	0.24
(B) Log 10 pathogen load						
Model						
LZ + pCO$_2$	81.5	0.46				
LZ $\times$ pCO$_2$	82.1	0.35				
LZ	83.2	0.19				
Weighted average model						
Intercept			0.25	0.26	1.0	0.33
LZ			**1.76**	**0.34**	**4.9**	**<0.0001**
High pCO$_2$			−0.41	0.36	1.1	0.26
LZ $\times$ high pCO$_2$			−0.83	0.51	1.6	0.10

Notes: Best models (ΔAICc < 3) for eelgrass disease severity (percent of leaves with lesions) (a) and log10 pathogen loads, followed by the weighted average of the top models (b). Model weights are given next to AICc values. Significant effects ($P < 0.05$) are in bold.

systems (e.g., *Vibrio* spp. in mussels and blood clams, Asplund et al. 2014, Zha et al. 2017). Collectively these limited results suggest that relatively small changes in OA can influence marine diseases in various ways that are not yet understood or predictable and more research on this topic warranted.

Table 4. Phase II results.

	AICc	Weight	Estimate	Adjusted SE	SE	df	*z*-value	*t*-value	*P*-value
(A) Phenolic compounds									
Model									
LZ + $p\mathrm{CO}_2$	−21.2	0.63							
$p\mathrm{CO}_2$	−19.0	0.20							
LZ × $p\mathrm{CO}_2$	−18.6	0.17							
Weighted average model									
Intercept			**1.107**	**0.089**			**12.1**		**<0.0001**
LZ			**0.236**	**0.089**			**2.5**		**0.01**
High $p\mathrm{CO}_2$			**−0.251**	**0.979**			**2.5**		**0.01**
LZ × High $p\mathrm{CO}_2$			−0.129	0.151			0.8		0.42
(B) Condensed tannins									
Best fit model									
Intercept			**0.234**		**0.009**	**26.0**		**24.8**	**<0.0001**
LZ			0.012		0.014	13.1		1.1	0.29
High $p\mathrm{CO}_2$			0.005		0.014	26.7		0.3	0.71
LZ × High $p\mathrm{CO}_2$			**−0.067**		**0.017**	**13.6**		**−4.0**	**0.002**

Notes: Best models (ΔAICc < 3) for concentrations (% dry mass) of phenolic compounds (a) and condensed tannins (b), followed by the weighted average of the top models. Since there was only one best-fitting model for condensed tannins, only the results of this model are shown. Significant effects ($P < 0.05$) are in bold.

The effects of oysters on EWD shoot severity differed between phases I and II. Juvenile oysters reduced EWD shoot severity in phase I, but not in phase II of the experiment. The high concentration of infectious LZ cells used in phase II may explain this difference. The dosage (60,000 cells per ml) of pathogen may have been so concentrated that filtration by 15 juvenile oysters could not substantially reduce the concentration of LZ prior to transmission in phase II. In contrast, during phase I, where EWD resulted from natural infections that occurred prior to collection, oysters reduced EWD shoot severity. In this case, the juvenile oysters may have been able to depurate the low concentrations of pathogenic cells that were produced by infected eelgrass and would have otherwise caused additional infections and visible lesions on the plants.

Filtration rates were recently modeled using 25.4 mm long oysters and averaged ~9 $\mathrm{ml}\cdot\mathrm{min}^{-1}\cdot\mathrm{oyster}^{-1}$ at 12°C and ~11 $\mathrm{ml}\cdot\mathrm{min}^{-1}\cdot\mathrm{oyster}^{-1}$ at 15.0°C (equation 11 in Ehrich and Harris 2015). Using this rate for phase I, the 32 oysters in each tank could filter seawater at a total rate of ~290 mls/min (turnover rate of once every 12 min or 5 times per hour), while the flowing seawater rate of 60 mls/min resulted in one tank turnover per hour. At 15°C, the 15 oysters in phase II trials would filter the entire tank volume every 21 min, or nearly three times the rates of inflow water exchange. However, Ehrich and Harris (2015) also observed a significant decrease in filtration rate as particle concentrations increased. Thus the oysters in our trials had the capacity to filter the entire volume of tank water 3–5 times faster than the tank turnover time demonstrating their ability to influence LZ levels when the pathogen concentrations did not inhibit filtration and could account for the differences in influence of oysters on EWD between phases I and II.

Further investigation is necessary to test whether oysters are ingesting LZ, and what effect ingestion has on the viability of LZ. LZ transmission could also be reduced if the pathogen is not actually ingested but is, instead, adhering to the oyster shell (e.g., *Himasthla elongate* on Pacific oyster shell, Welsh et al. 2014) or expelled in aggregated pseudofeces, which may compromise their pathogenicity (reviewed in Burge et al. 2016). Regardless of the mechanism, our results suggest oysters may be useful for controlling concentrations of protistan pathogens found in natural environments. These results add to a growing body of literature suggesting that filtration of a variety of water-borne micro and macro parasites (e.g., avian influenza and sea lice) by filter-feeders may alter epidemiological patterns (Faust et al. 2009, Webb et al. 2013, Bidegain et al. 2016, Burge et al. 2016). Field-based studies are required to further understand how this mechanism functions in more natural systems.

Some control eelgrass ramets in phase II (not experimentally exposed to LZ) showed signs of mild shoot severity, which were significantly less severe than those in the LZ treatments. The mild EWD in the control plants in is likely due to background infections carried over since collection in the wild. The near absence of LZ in the control treatments (from highly sensitive qPCR assays) further validates our conclusion that the lesions found in control plants resulted from very light carry-over infections from the field, which would be randomly distributed across treatments.

The positive effect of increased $p\mathrm{CO}_2$ on eelgrass and the negative effect of increased $p\mathrm{CO}_2$ on oyster mass are consistent with previous studies (Jiang et al. 2010, Barton et al. 2012, Koch et al. 2013, Zimmerman et al. 2017). Increased growth in eelgrass resulting from elevated $p\mathrm{CO}_2$ has been found in other seagrass species when resources such as light and nutrients are not limiting photosynthetic rates (Jiang et al. 2010, Alexandre et al. 2012, Koch et al. 2013). While the presence of eelgrass increased the daytime pH of the seawater by nearly 0.3 in the high $p\mathrm{CO}_2$ treatment, this did not ameliorate the negative effects of a 3-week exposure to high $p\mathrm{CO}_2$. It is possible that, in the eelgrass-oyster co-culture treatment, the lower nighttime pH counteracted any positive effects of the higher daytime pH levels on oyster growth. Nonetheless, the detectable difference in mass of juvenile oysters in the high $p\mathrm{CO}_2$ treatments shows that relatively short (i.e., 3-week) exposure to high $p\mathrm{CO}_2$ conditions can reduce juvenile oyster growth. Negative impacts of OA on the survival and development of larval oysters are well documented in Pacific oysters (Barton et al. 2012) and our data supports growing evidence that OA slows the growth of juvenile oysters (i.e., Lannig et al. 2010, Kroeker et al. 2013).

Further experimentation is necessary to quantify if and when the phytoremediation of seawater chemistry that occurred in this laboratory experiment scales to more natural systems. Numerous biological and environmental factors can alter phytoremediation of pH, including eelgrass density, water flow, mesograzers and epiphytes (Connell and Russell 2010, Alsterberg et al. 2013, Hughes et al. 2018). Moreover, the amplitude of the diel pH cycling due to seagrasses is predicted to increase in the future as a result of OA and coastal acidification (Salisbury et al. 2008, Melzner et al. 2013, Cyronak et al. 2018). How oysters and other OA-sensitive organisms respond to diel variation in pH is likely context-dependent; for example, a recent study on *C. virginica* in the Chesapeake Bay found that effects on growth depended not only on Ω_{CAL}, but also on dissolved oxygen concentrations (Keppel et al. 2016). Diel cycling of pH in the Salish Sea region of the Northeast Pacific is less well documented and it is not known how Pacific oysters respond to increasing amplitude diel pH cycles.

Total phenolic compounds in eelgrass ramets were less concentrated in response to high pCO_2 and more concentrated in response to LZ exposure. Condensed tannins were less concentrated when eelgrass ramets were exposed to both high pCO_2 and LZ. The lower concentrations of total phenolic compounds that we documented in eelgrass (*Z. marina*) in response to high pCO_2 are consistent with studies with the seagrass *Z. muelleri*, which showed a decrease in total phenolic concentrations when grown in high pCO_2 waters (Arnold et al. 2014) and has been attributed to decreased carbon uptake or a reallocation of carbon to other pathways. While increases in concentrations of total phenolic compounds in eelgrass with EWD have been found previously (i.e., Vergeer and Develi 1997, Groner et al. 2016), the role of phenols in fighting LZ infections is still unclear (i.e., Buschbaum et al. 1990, Vergeer and Develi 1997). The functional role of condensed tannins as a defense against EWD is also unclear. Further investigation into which specific phenolic compounds are being made and their effect on LZ growth in eelgrass would aid in understanding whether the changes in phenolic compounds found in response to LZ exposure are adaptive.

In conclusion, our study provides preliminary evidence to suggest that eelgrass and possibly oysters could benefit from co-culture under projected pCO_2 conditions. The use of Pacific oysters for aquaculture is increasing (FAO 2014), while seagrasses are declining at an estimated rate of 7% per year, despite extensive restoration efforts (Waycott et al. 2009). Directly remediating disease threats to seagrasses by increasing nearby oyster populations may improve seagrass health; however, further research is necessary to optimize culture conditions and explore these effects. For example, high densities of oysters and suspended oyster frames have been found to be detrimental to seagrasses (Kelly and Volpe 2007, Wagner et al. 2012). We suggest that future research should evaluate these interactions in natural settings and focus on the direct and indirect influences of potentially co-occurring environmental changes such as alterations to seawater chemistry and temperature.

ACKNOWLEDGMENTS

This research was funded through the Canadian Excellence Research Chair in Aquatic Epidemiology at the University of Prince Edward Island (to MLG and RC), start-up funds provided to CAB from the University of Maryland Baltimore County and the University of Maryland Baltimore, Washington Sea Grant project R/LME-5 to CSF and School of Aquatic and Fishery Sciences at the University of Washington, and National Science Foundation grant IOS-00100768 to KLV. Michelle Herko, Rebecca Guenther, Constance Sullivan and Reyn Yoshioka assisted in the execution of the experiment.

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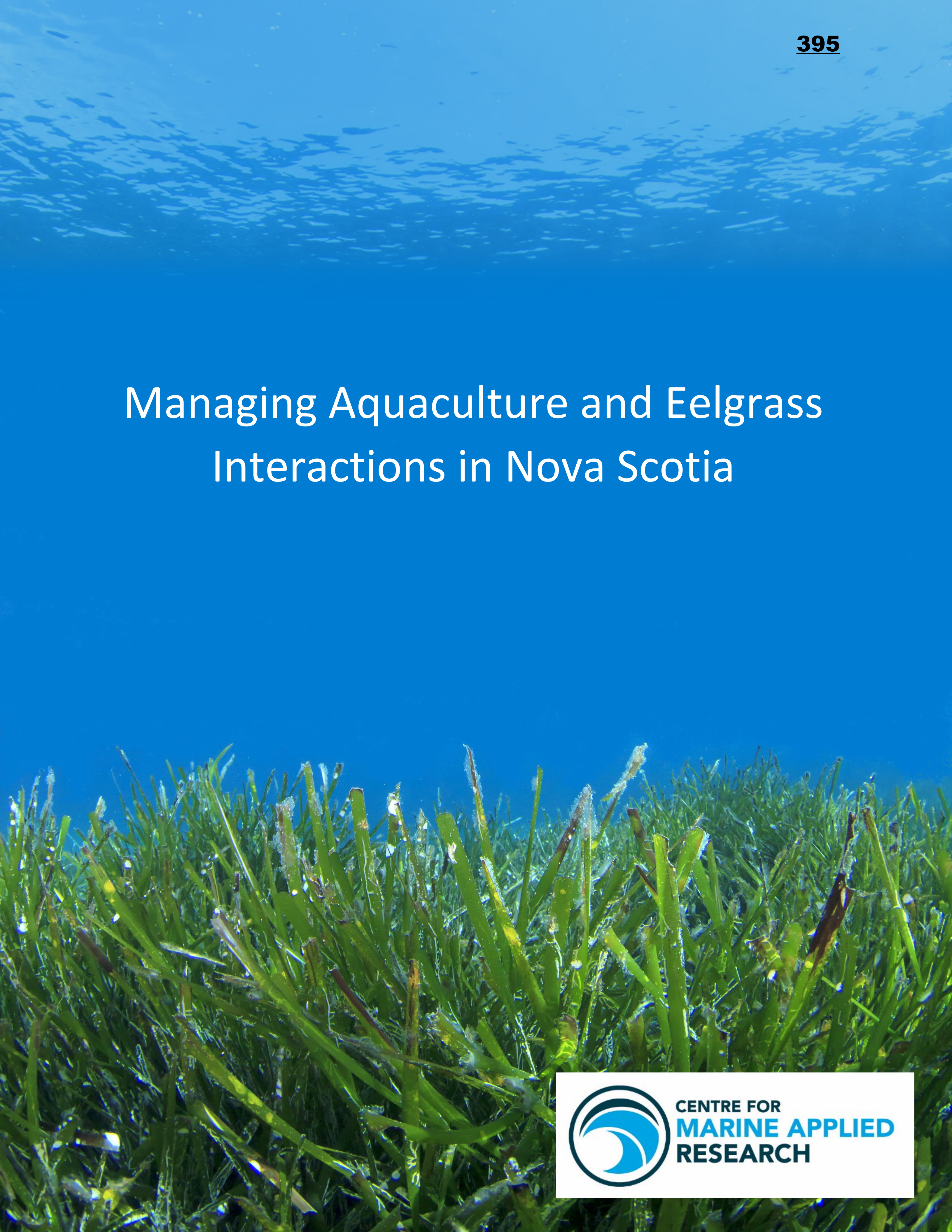
Supporting Information

Additional supporting information may be found in the online version of this article at http://onlinelibrary.wiley.com/doi/10.1002/ecy.2393/suppinfo

Data Availability

Data and statistical code are available on figshare at https://doi.org/10.6084/m9.figshare.6182522.



Managing Aquaculture and Eelgrass Interactions in Nova Scotia

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RESEARCH

Managing aquaculture and eelgrass interactions in Nova Scotia

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Suggested Citation:
Howarth, L.M., Lewis-McCrea. L., LaBelle, J., and Reid, G.K., (2021) Managing Aquaculture and Eelgrass Interactions in Nova Scotia. Centre for Marine Applied Research (CMAR), Dartmouth, Nova Scotia, Canada. 53 pages.

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Executive summary

Eelgrass (*Zostera marina*) is the primary seagrass species in Atlantic Canada. Eelgrass meadows are ecologically important as they can provide food and habitat to a wide range of birds, fish, and invertebrates. Eelgrass can also enhance sediment stability, improve water clarity, and protect coastlines from erosion. Consequently, eelgrass is an 'Ecologically Significant Species' (ESS) and protected under federal legislation through a prohibition on the harmful alteration, disruption or destruction (HADD) of fish habitat.

Some eelgrass beds in Atlantic Canada have receded in recent years due to a multitude of interacting stressors including disease, species invasions, nutrient enrichment, and climate change. There have been concerns that aquaculture may also have the potential to negatively impact eelgrass, given aquaculture is primarily a coastal activity. This report was written by the Centre for Marine Applied Research (CMAR) to review the potential effects of shellfish and finfish aquaculture on eelgrass beds in Nova Scotia.

Most studies on the impacts of finfish farms on seagrass have examined two species of Neptune grass (*Posidonia oceanica* and *Cymodocea nodosa*) in the Mediterranean Sea. These studies report a general decrease in seagrass cover with increasing proximity to finfish farms for distances up to 300 m, primarily due to the deposition of particulate organic wastes. However, these studies may have limited relevance to finfish aquaculture facilities in Nova Scotia. This is because Mediterranean fish farms are often situated in low nutrient ('oligotrophic'), low energy environments in shallow depths, very close to shore. To date, only one field study has investigated finfish aquaculture and seagrass interactions outside of the Mediterranean, which studied a finfish farm in Port Mouton Bay, Nova Scotia. This study found some evidence of eelgrass cover declining with increasing proximity to the finfish farm, but overall, trends were less clear than those reported in the Mediterranean. Further investigation is warranted as a single field study is insufficient to reach definitive conclusions on finfish aquaculture / seagrass interactions in temperate ecosystems.

In comparison to finfish aquaculture, the potential effects of shellfish aquaculture on eelgrass are better documented. Studies show the primary impact on seagrass is shading from aquaculture gear and infrastructure. Correspondingly, any negative effects are usually highly localized. In general, suspended shellfish aquaculture has less potential to impact eelgrass compared to on-bottom methods.

Managing eelgrass and environmental impact interactions can be difficult as regulators typically have access to very little data on eelgrass, and do not have the resources available to perform detailed surveys on a large scale. However, proposed aquaculture operations must undergo baseline monitoring during which the presence of eelgrass and other fish habitat are assessed by the Federal Government. Mitigation or avoidance measures are then imposed if the operation is considered to pose a risk to fish habitat.

The issue of potential eelgrass / aquaculture interactions has largely been addressed in Eastern New Brunswick due to the adoption of a comprehensive Bay Management Framework (BMF) system for suspended oyster aquaculture. The BMF established a broad range of site selection criteria and operating conditions for the suspended oyster aquaculture industry, which help to ensure oyster aquaculture has minimal impacts on eelgrass without requiring additional data collection. Based on the success of the BMF, a similar management system could be adapted for Nova Scotia. However, the BMF only address the potential impacts of suspended oyster aquaculture. This report proposes several additional measures for other gear types, as well as a consultation process, which could help minimize aquaculture impacts on eelgrass in Nova Scotia.

1. Introduction

1.1. What is seagrass?

Seagrasses are grass-like flowering plants (or 'angiosperms') which grow in areas that are tidally or fully submerged by seawater. They can form dense aggregations known as 'meadows' or 'beds' (Figure 1), and are widely distributed across tropical and temperate coastlines, estuaries and lagoons (Short et al. 2007, Dinusha and Costello 2018). There are around 60 seagrass species worldwide, belonging to 11 different genera and 4 families (Dinusha and Costello 2018). As they have no special morphological or genetic characteristics distinguishing them from other aquatic plants, seagrasses form an ecological group, not a taxonomic group (Tomlinson 1982, den Hartog and Kuo 2006).

Figure 1 | Eelgrass (*Zostera marina*) meadows growing off the coast of Nova Scotia. Source: Fisheries and Oceans Canada (DFO).

1.2. Eelgrass (*Zostera marina*)

There are a total of 11 seagrass genera (reviewed in Jacobs and Les 2009). The genus *Zostera* (family: Zosteraceae) consists of 15 seagrass species. Of these, *Zostera marina* or 'eelgrass' (Figure 1), is the most widespread species in the northern hemisphere of the Pacific and Atlantic Oceans (Green and Short 2003).

1.3. Distribution of eelgrass in Nova Scotia

Eelgrass is the primary seagrass species on the eastern coast of Canada and USA (Fisheries and Oceans Canada 2009). As eelgrass occurs from North Carolina up to northern Quebec, its range includes parts of Hudson Bay, Newfoundland and Labrador, New Brunswick, Prince Edward Island (PEI) and Nova Scotia. In Nova Scotia, eelgrass has been recorded throughout the Northumberland Strait, around Cape Breton, and down to the south shore beyond Yarmouth (Figure 2). As eelgrass does not grow in areas of high energy and turbidity, it tends to be scarcer in the Bay of Fundy (Moore and Short 2006, Fisheries and Oceans Canada 2009, Murphy et al. 2020).

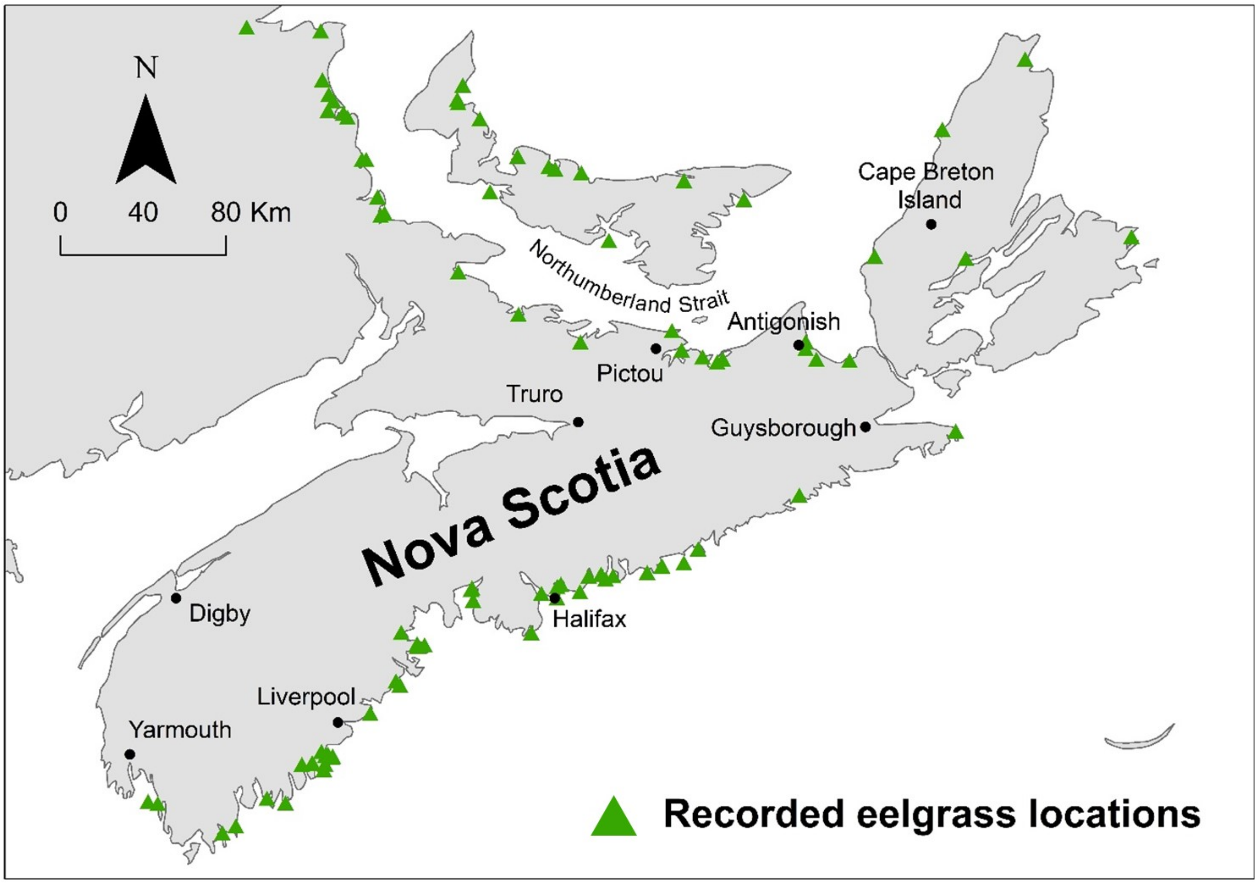

Figure 2 | Locations of recorded eelgrass observations according to data collated by Environment Climate Change Canada (ECCC 2021). These data are not comprehensive and are highly unlikely to reflect the exact present distribution of eelgrass.

Although eelgrass meadows have been reported throughout Nova Scotia and Atlantic Canada, their exact location, density, and overall health are largely unknown. Consequently, Fisheries and Oceans Canada (DFO) have established a 'National Eelgrass TaskForce' (NETForce) which aims to compile all available eelgrass data from across Canada and create a national eelgrass map by April 2022. NETForce are also in the process of satellite mapping eelgrass beds in Nova Scotia (e.g. Wilson et al. 2020) and developing species distribution models (Melisa Wong, DFO, pers. comm. 9th July 2020). DFO's Marine Planning and Conservation unit for the Gulf region are also satellite mapping eelgrass in the Gulf of St Lawrence (Jeffrey Barrell, DFO, pers. comm. 29th September 2020).

1.4. Environmental criteria suitable for eelgrass

There are a wide range of chemical, biological and physical parameters (Table 1) that can influence the distribution of eelgrass (Fisheries and Oceans Canada 2009). There are also many stressors and disturbances that can alter these parameters (see Section 5), which can have significant effects on eelgrass and the ecological communities they support (see Section 3).

Table 1 | Key environmental parameters that can affect eelgrass distribution.

Parameter	Thresholds	References
Ammonium (NH_4^+)	Aquatic toxicity begins at 25 µM and mortality occurs at 125 µM.	van Katwijk et al. (1997)
Current speed	Can tolerate a range of 16 – 180 cm s^{-1}.	Fisheries and Oceans Canada (2009)
Dissolved oxygen (O_2)	Minimum of 2.02 mg O_2 L^{-1} in water.	Fisheries and Oceans Canada (2009)
Hydrogen sulphide (H_2S)	Sediment toxicity begins at 100 µM and mortality occurs at 680 µM.	Fisheries and Oceans Canada (2009), Dooley et al. (2013)
Light	Minimum light requirement: 11 – 34 % surface irradiance (SI) or 1.2 – 12.6 mol photons m^{-2} day^{-1}.	van Katwijk et al. (1997), Hauxwell et al. (2003), Eriander (2017), Bertelli and Unsworth (2018)
Nitrate (NO_3^-)	Aquatic toxicity effects begin at 35 µM and mortality occurs at ~ 250 µM.	Burkholder et al. (1992)
Salinity	Optimal range: 20 – 26 ppt Tolerable range: 5 – 35 ppt	Fisheries and Oceans Canada (2009)
Sediment composition	Reported in sediments ranging in particle size from mud to cobbles.	Fisheries and Oceans Canada (2009)
Redox potential of sediment	Tolerable range for seagrasses in general: -175 to +300 mV.	Marbá et al. (2006)
Water temperature	Optimal range: 10 – 25 °C Tolerable range: 0 – 35 °C	Fisheries and Oceans Canada (2009)
Water depth	The euphotic zone. Maximum of 12 m, but usually occurs between 1 – 7 m.	Moore and Short (2006), (Murphy et al. 2020), Jeffrey Barrell and Melisa Wong (DFO, pers. comm)

2. Eelgrass ecology and life history

2.1. Life history

Eelgrass, like all seagrasses, are clonal plants that grow by replicating modules (or ramets) along their rhizome (Figure 3). These modules consist of: (1) a shoot, which extends into the water column and bears photosynthetic leaves; (2) roots, which anchor the plant in the sediment; and (3) a segment of rhizome, which connects to neighboring modules (Reviewed in Duarte et al. 2006). New modules are formed along the rhizome as it grows and extends horizontally through the sediment, allowing the plant to expand into new areas. Over time, some modules may become physically separated by disturbance events (see Section

5), or through the natural senescence of shoots, resulting in multiple individuals that are all genetically identical. Consequently, although eelgrass meadows appear to consist of many individual plants, they may be connected to the same rhizome, and even if they are not physically connected, they may be genetically identical (Waycott et al. 2006).

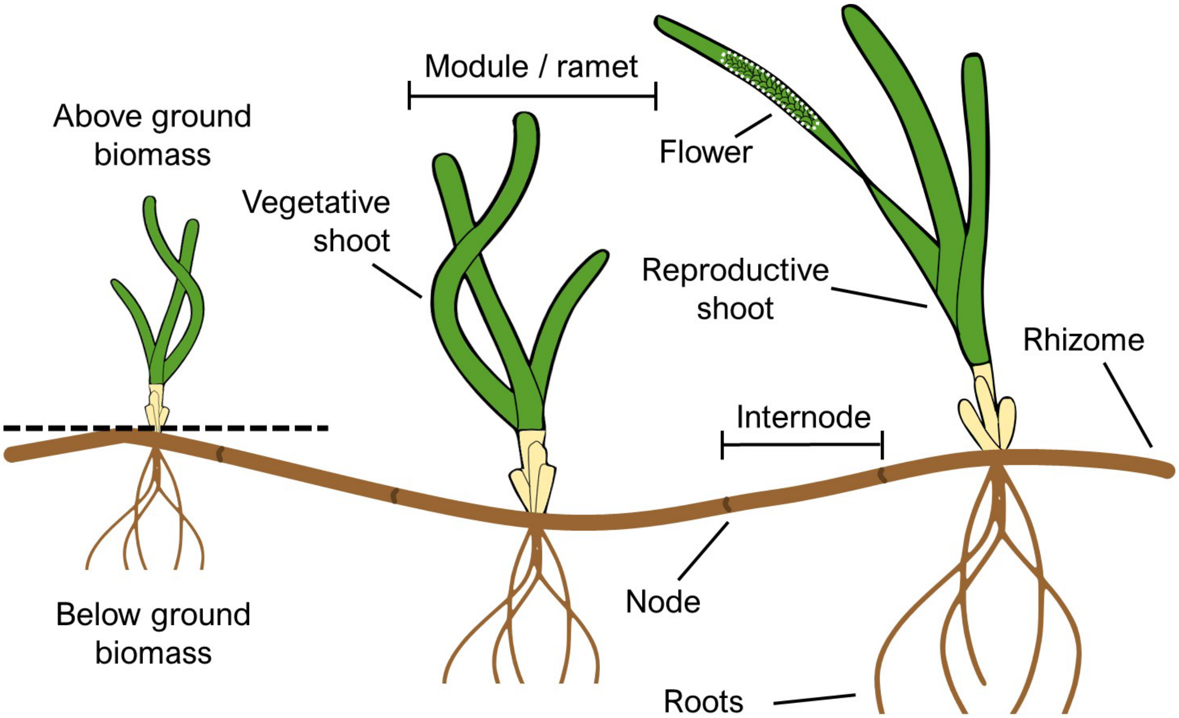

Figure 3 | Key morphological structures of a single eelgrass (*Zostera marina*) plant. Typically, leaves are 20 – 50 cm long but can reach up to 1.5 m; the roots can reach 20 cm in length; the rhizome is 2 – 6 mm thick; and the internodes can range from 5 – 40 mm in length (Borum and Greve 2004, Duarte et al. 2006).

Eelgrass shoots can be vegetative or reproductive. Reproductive shoots are generally taller, more branched, bushier in appearance, and encapsulate male and female flowers along the mid-length of their blades (Tomlinson 1982, Borum and Greve 2004). Male flowers release linear strands of pollen to pollinate the female flowers of neighboring plants. If female flowers become fertilized by the pollen, seeds will form and mature over a period of a few weeks (Figure 4). These seeds are released directly into the water column where they typically become dispersed over 1 – 10 m before settling on the seabed (Reviewed in Marbá et al. 2004). Alternatively, the entire flower or reproductive shoot can break off and float away, dispersing the seeds over a much larger distance (Marbá et al. 2004, Moore and Short 2006). Eelgrass seeds rarely survive longer than a year and may remain dormant within the sediment for up to 6 months before germinating (Coolidge Churchill 1983, Orth et al. 2000). This pool of viable seeds is often referred to as a population's 'seed bank' (Harwell and Orth 2002, Duarte et al. 2006). Sexual reproduction and the resulting seed bank can play an important role in recovery processes because, even if a disturbance destroyed all above and below ground biomass of an eelgrass community (see Section 5), the seed bank may still allow the community to recover in the future (Harwell and Orth 2002).

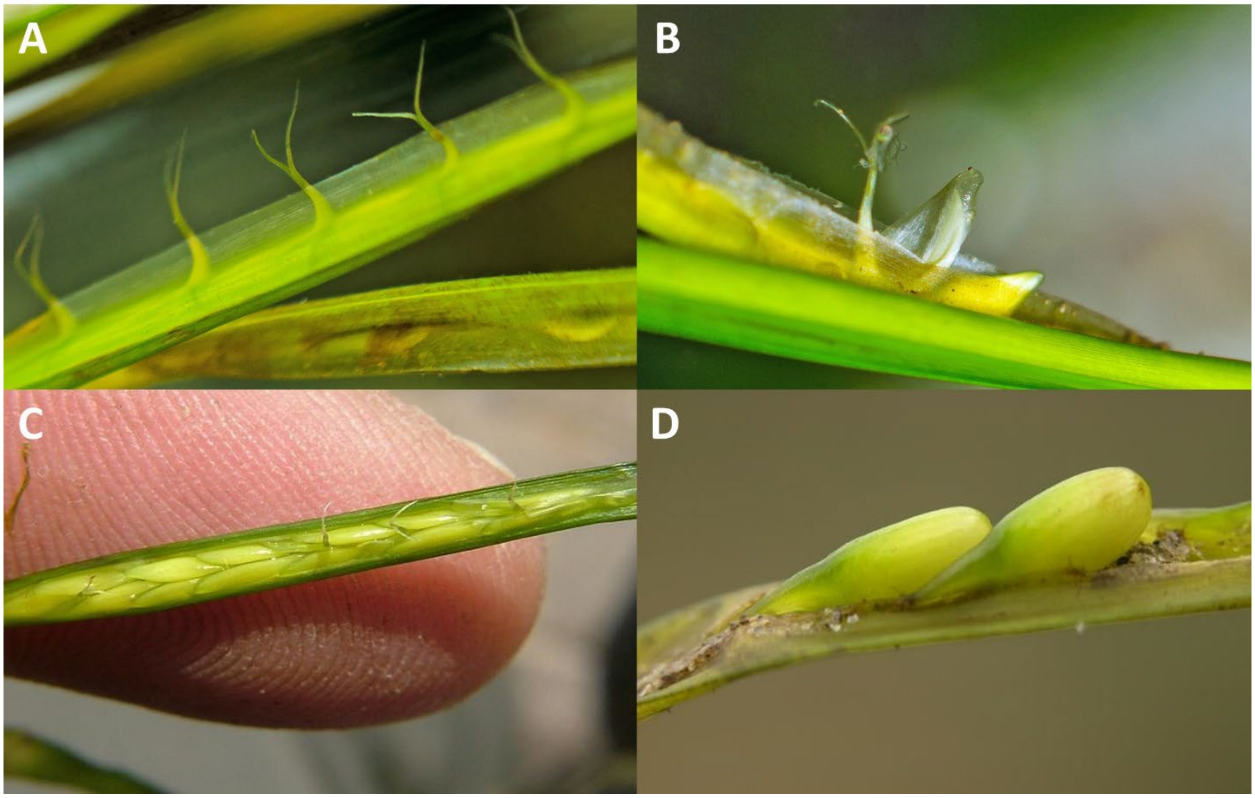

Figure 4 | Reproductive structures of eelgrass: (A) female flower; (B) male flower; (C) developing seeds; (D) seeds ready for dispersal. Source: Images A & B courtesy of Susannah Anderson (https://wanderinweeta.blogspot.com); Images C & D courtesy of David Fenwick Sr (www.aphotomarine.com).

Most seagrasses exhibit very low rates of sexual reproduction and seed production, as the proportion of shoots that flower is usually lower than 10 % (Durako and Moffler 1985, Duarte et al. 1997, Marbà and Walker 1999, Campey et al. 2002). However, sexual reproduction and seed production can exhibit remarkably high levels of spatial and temporal variation (Duarte et al. 2006, van Katwijk et al. 2010). In addition, sexual reproduction has been shown to increase in response to stressors and disturbances, such as fishing disturbance, heat waves, freshwater influxes and storms (reviewed in Cabaço and Santos 2012). Presumably, this is an evolutionary response to promote recovery from the seed bank.

Eelgrass generally has a perennial life history. Under this strategy, plants live for multiple years and reproduction primarily occurs through asexual clonal / vegetative growth (Olesen 1999). However, eelgrass populations employing an annual life history have been reported on several coasts in the Pacific and Atlantic, including in Petpeswick, Nova Scotia (Keddy and Patriquin 1978, Meling-López and Ibarra-Obando 1999, van Katwijk et al. 2010). These annual plants die during the winter but then re-establish from the seed bank, meaning annual plants can complete their life cycle in less than 12 months (Jarvis et al. 2012). In contrast, the seedlings of perennial plants generally require 1 – 2 years before they are able to start flowering (Keddy and Patriquin 1978). Annual populations invest substantially more energy into sexual reproduction than perennial plants, generating an average of 24,000 seeds per square metre compared to 6,200 seeds per square metre in perennial populations (Olesen 1999). The ability for annual populations to die and re-establish from the seed bank is thought to provide a mechanism to mitigate seasonal disturbances such as warm summer temperatures or scouring by sea ice (Keddy and Patriquin 1978, Robertson and Mann 1984, Santamaría-Gallegos et al. 2000). Interestingly, the seeds of perennial plants can give rise to both annual and perennial plants and vice versa (Keddy and Patriquin 1978), and some eelgrass populations are comprised of both annual and perennial plants, known as 'semi-annuals' or 'mixed annuals' (van Katwijk et al. 2010, Jarvis et al. 2012, Vercaemer et al. in press).

Differences in environmental conditions can cause eelgrass morphology to vary within and between populations. For example, the ratio of above ground to below ground biomass can be affected by local hydrodynamic conditions, as eelgrass in high energy environments tend to invest more energy in below ground structures (i.e. roots and rhizomes), presumably to help prevent dislodgement (reviewed in Peralta et al. 2007). Likewise, light limitation can cause eelgrass to display a reduction in below ground biomass and an increase in leaf length and / or shoot density to maximize photosynthetic capability / tissues, and to reduce the amount of stored carbohydrates plants need to allocate to root tissues (Olesen and Sand-Jensen 1993). Overall, a range of biotic (e.g. grazing, and inter- and intra-specific competition) and abiotic (nutrients, temperature and salinity) stressors and disturbances can affect eelgrass morphology and physiology (see Section 5).

2.2. Patch dynamics

Some seagrass populations can form continuous meadows that stretch for hundreds of kilometres (Carruthers et al. 2007), while others may naturally occur as a series of highly fragmented patches (Duarte et al. 2006). Patchy seagrass cover can be the product of colonization processes. For example, new patches can be established by seedlings, or by the dispersal of broken-off shoots which can re-anchor and resume growth in new locations (Marbá et al. 2004). Alternatively, shoot senescence or disturbance events (see Section 5) may cause the loss of some parts of the seagrass meadow, also resulting in fragmented cover (Bell et al. 2006). All these processes can promote the development of a mosaic of patches of different ages and developmental stages (Duarte and Sand-Jensen 1990, Olesen and Sand-Jensen 1994, Vidondo et al. 1997). Consequently, although seagrass meadows appear to be static, they are highly dynamic landscapes in a constant state of contraction, expansion, recruitment, and mortality (Duarte et al. 2006, Marbá et al. 2006, Walker et al. 2006). These high levels of natural variability can complicate monitoring efforts and make it difficult to determine how seagrass meadows respond to disturbance (also see Section 5.7).

2.3. Seasonal fluctuations

Seagrasses can exhibit strong variations in growth, reproduction, and biomass in response to seasonal fluctuations in water temperature and light availability. These fluctuations tend to be more pronounced in temperate and high-latitude seagrass communities as these regions experience stronger seasonal changes in light and temperature than tropical and sub-tropical regions (Duarte et al. 2006).

Generally, eelgrass biomass and shoot density are highest during the summer, when warmer temperatures and greater light availability promote faster rates of growth (Kaldy and Lee 2007). For example, eelgrass beds in Chesapeake Bay (east coast, USA) tend to exhibit greater biomass, leaf length and shoot density between June – July (Orth and Moore 1986). Eelgrass then undergo extensive leaf loss between July – August when water temperatures exceed 25 °C, corresponding with a 200 – 400 % reduction in shoot density and above and below ground biomass. Biomass and shoot density then remain supressed throughout September – April until temperatures begin to rise again during the spring. Chesapeake Bay also exhibits seasonal increases in turbidity which can negatively impact eelgrass recruitment and seedling survival (Moore et al. 1997). Field observations in Nova Scotia, Oregon (west coast, USA), and South Korea also show eelgrass can exhibit fluctuations in growth, biomass, shoot density, and shoot senescence, in response to seasonal changes in light and water temperature (Lee et al. 2005, Kaldy and Lee 2007, Kwak and Huh 2009, Wong et al. 2013). In contrast, several eelgrass beds in Pomquet Harbour and Chezzetcook Inlet, Nova Scotia, display annual fluctuations in biomass due to winter scouring by sea ice (Robertson and Mann 1984, Schneider and Mann 1991). Scouring may be widespread across Nova Scotia as winter ice occurs in potential eelgrass habitat throughout the province (Jeffrey Barrel, DFO, pers. comm. 29th September 2020).

3. The ecological significance of eelgrass

3.1. Sediment stability, biogeochemistry, and water clarity

Seagrasses are often described as "ecosystem engineers" for their ability to modify their physical, chemical, and biological environment (Jones et al. 1997, Bos et al. 2007). For example, as water currents and waves pass over seagrass meadows, some of their energy becomes dissipated. This reduction in water velocity can protect shorelines from coastal erosion and encourages sediment particles suspended in the water to settle on the seafloor (Ondiviela et al. 2014). Seagrasses prevent the resuspension of these sediments by trapping them within their root and rhizome networks (Koch et al. 2006), which can enhance sediment stability, improve water clarity, and allow more light to penetrate to deeper depths (Folkard 2005, Carr et al. 2010).

Seagrass beds can also trap detritus (e.g. dead leaves and rhizomes) and other organic matter, which can act as a carbon store (see Section 3.7). This input of organic matter can greatly boost microbial activity and lead to the formation of distinct bacterial communities compared to surrounding areas (Gacia and Duarte 2001, Marbá et al. 2006, Tarquinio et al. 2019). As bacteria play a fundamental role in ocean biogeochemistry, seagrass meadows can strongly influence the cycling of carbon, nitrogen, sulphur, phosphorus and oxygen (Marbá et al. 2006, Mateo et al. 2006, Romero et al. 2006, Liu et al. 2018).

3.2. Infauna and epifauna

As seagrass beds can improve water clarity, and increase the stability and organic content of sediments, they are often associated with diverse communities of epifauna (i.e. organisms living on seagrass or on the sediment) and benthic infauna (i.e. organisms living within the sediment). Several studies have observed that eelgrass meadows in the Northwest Atlantic support greater diversity and abundance of hydroids, bryozoans, gastropods, polychaetes, amphipods, and other invertebrates compared to non-eelgrass habitats (Orth 1973, Orth 1977, Fisheries and Oceans Canada 2009, Joseph et al. 2012, Wong 2018, Wong and Kay 2019). Likewise, eelgrass beds in Atlantic Canada are often associated with a higher density and greater diversity of seaweeds and epiphytic algae, of which more than 20 species are dependent on eelgrass to complete their lifecycle (Fisheries and Oceans Canada 2009, Schmidt et al. 2012). These organisms can contribute to the food web by providing food to a variety of larger animals including crustaceans, fish, and birds (see Sections 3.3 – 3.9).

3.3. Greater food availability and growth rates

The plant and invertebrate communities associated with seagrass beds can provide food to a wide range of organisms. Correspondingly, eelgrass beds across the Northwest Atlantic coast have been shown to support faster growth rates of juvenile Atlantic cod (*Gadus morhua*), white hake (*Urophycis tenuis*), cunner, (*Tautogolabrus adspersus*), blue crab, (*Callinectes sapidus*) and tautog (*Tautoga onitis*) (Tupper and Boutilier 1995, 1997, Heck et al. 2003, Renkawitz et al. 2011). However, these benefits are not universal as Greenland cod (*Gadus ogac*), winter flounder (*Pseudopleuronectes americanus*) and naked goby (*Gobiosoma bosci*) have been shown to display slower growth rates in eelgrass beds compared to pelagic and unvegetated habitats, suggesting some species may face a trade-off between reduced predation risk (see Section 3.4) and greater food availability (Sogard 1992, Heck et al. 2003, Renkawitz et al. 2011). Likewise, a meta-analysis of over 200 papers showed that other structured habitats, such as macroalgae beds, kelp forest, and oyster and cobble reefs, can benefit juvenile growth rates just as much as seagrass meadows (Heck et al. 2003).

3.4. Predator refuge

Seagrass canopies provide three-dimensional structure to the seabed. In doing so, they can reduce the visual and swimming capabilities of predators, thereby providing a refuge to smaller organisms. For example, aquarium studies have shown predation rates on the daggerblade grass shrimp (*Palaemonetes pugio*) and juvenile Atlantic cod are lower within high densities of artificial eelgrass (Joseph et al. 2012). Similarly, a field study in Newfoundland showed juvenile Atlantic cod experienced lower predation risk in large eelgrass meadows compared to smaller patches and unvegetated areas (Gorman et al. 2009).

3.5. Nursery habitat

By offering greater food availability and protection from predators, seagrass meadows can provide 'nursery habitat' to a wide range of juvenile fish and crustaceans, many of which are of commercial importance (Heck et al. 2003, Bertelli and Unsworth 2014, 2018). Field studies across Atlantic Canada have shown eelgrass beds support higher abundances of juvenile fish including Atlantic cod, cunner, white hake, mummichog (*Fundulus heteroclitus*), Atlantic silversides (*Menidia menidia*), northern pipefish (*Syngnathus fuscus*), as well as fourspine (*Apeltes quadracus*) and threespine (*Gasterosteus aculeatus*) sticklebacks (Gotceitas et al. 1997, Laurel et al. 2003, Joseph et al. 2006, Grant et al. 2007, Renkawitz et al. 2011, Joseph et al. 2012, Schein et al. 2012, McCain et al. 2016). Some of these species (e.g. pipefish and sticklebacks) are known to spawn in eelgrass beds, making them important habitats for their reproduction (Schein et al. 2012).

3.6. Aquatic birds

Eelgrass, and their associated invertebrate and algae communities, form an important dietary component for several migratory birds in Atlantic Canada including American black ducks (*Anas rubripes*), Atlantic brant (*Branta bernicla*), Barrow's goldeneye (*Bucephala islandica*), Canada geese (*Branta canadensis*) and common goldeneye (*Bucephala clangula*) (Hanson 2004a). Of these, Atlantic brant and Canada geese are known to feed almost exclusively on eelgrass shoots (Erskine 1997, Martell 1997, Newman-Smith 1997, Ganter 2000). In some cases, the links between eelgrass meadows and migratory birds are well established. For example, when eelgrass beds in the Antigonish estuary, Nova Scotia, experienced a 95 % loss in below ground biomass between 2000 – 2001 (see Section 5.2), it was immediately followed by a 50 % reduction in goldeneye abundance and the near disappearance of Canada geese from this region (Seymour et al. 2002). Similarly, the dramatic loss of eelgrass from wasting disease in the early 1930's (see Section 5.1) led to the fall migration pattern of Atlantic brant to no longer include a route along the coast of New Brunswick and Nova Scotia (Hanson 2004a). It has therefore been argued that any future declines in eelgrass would have major impacts on waterfowl feeding behaviour, migration patterns and over-winter survival (Seymour et al. 2002, Hanson 2004a, Fisheries and Oceans Canada 2009).

3.7. Blue carbon

Seagrass beds can sequester carbon into underlying sediments by trapping detritus within their rhizome networks. Consequently, seagrass and other forms of 'blue carbon' (e.g. mangroves) could help mitigate the effects of increasing global carbon dioxide levels and climate change (reviewed in Bedulli et al. 2020). However, studies show carbon storage within seagrass beds is highly variable, and in some cases, their carbon storage abilities may be no different than non-seagrass habitats (reviewed in Ricart et al. 2020).

3.8. Trophic subsidies

Seagrass and their associated communities provide energy, or 'trophic subsidy', to a wide range of organisms and ecosystems (see reviews by Mateo et al. 2006, Heck et al. 2008). For instance, seagrass detritus can provide a continual supply of organic matter to deep-sea ecosystems. Likewise, seagrass detritus can wash up on the shore in huge quantities, providing habitat and food to invertebrates, birds, and mammals. Lastly, the high densities of invertebrates and juvenile fish (see Section 3.5) associated with seagrass meadows can disperse into neighbouring habitats and contribute towards commercial fisheries.

4. Legislation regarding eelgrass in Canada

4.1. Ecologically Significant Species (ESS)

DFO designated Eelgrass as an 'Ecologically Significant Species' (ESS) in acknowledgement of its unique influence on the ecology of sand and mud flats in Canada (Fisheries and Oceans Canada 2009). This designation formally recognizes that if "eelgrass were to be perturbed severely, the ecological consequences would be substantially greater than an equal perturbation of most other species associated with this community". Although, ESS designations do not impose legal protection, it is intended to bring attention to species of high ecological significance, in order to promote a greater degree of risk aversion management regarding any human activities that may impact them, or their community properties (Fisheries and Oceans Canada 2007, Coll et al. 2011).

4.2. Marine Protected Areas (MPAs)

Marine Protected Areas (MPAs) aim to protect organisms and their habitats by partially or fully restricting human impacts within their boundaries. There are currently no MPAs in Atlantic Canada which specifically protect eelgrass beds as part of their management objectives. However, the Basin Head MPA, located on the eastern shore of PEI, encompasses some patches of eelgrass (Fisheries and Oceans Canada 2016a, b). Most of this eelgrass occurs within Zone 2 (Figure 5), which is less protected than Zone 1, as it permits small levels of oyster harvesting within its boundaries. Nonetheless, there are some general signs of eelgrass recovery within the MPA (Jeffrey Barrell, DFO, pers. comm. 16th July 2020).

The Eastern Shore Islands are located on the eastern shore of Nova Scotia and have recently been selected as an Area of Interest (AOI) by DFO (Fisheries and Oceans Canada 2020d). This 2,000 km^2 region (Figure 6) contains over 340 km^2 of eelgrass in its near-shore coastal areas (Wilson et al. 2020). If approved, the Eastern

Shore Islands AOI would be the first MPA in Atlantic Canada that officially lists the protection of eelgrass habitats as one of its management objectives (Tanya Koropatnick, DFO, pers. Comm, February 2021).

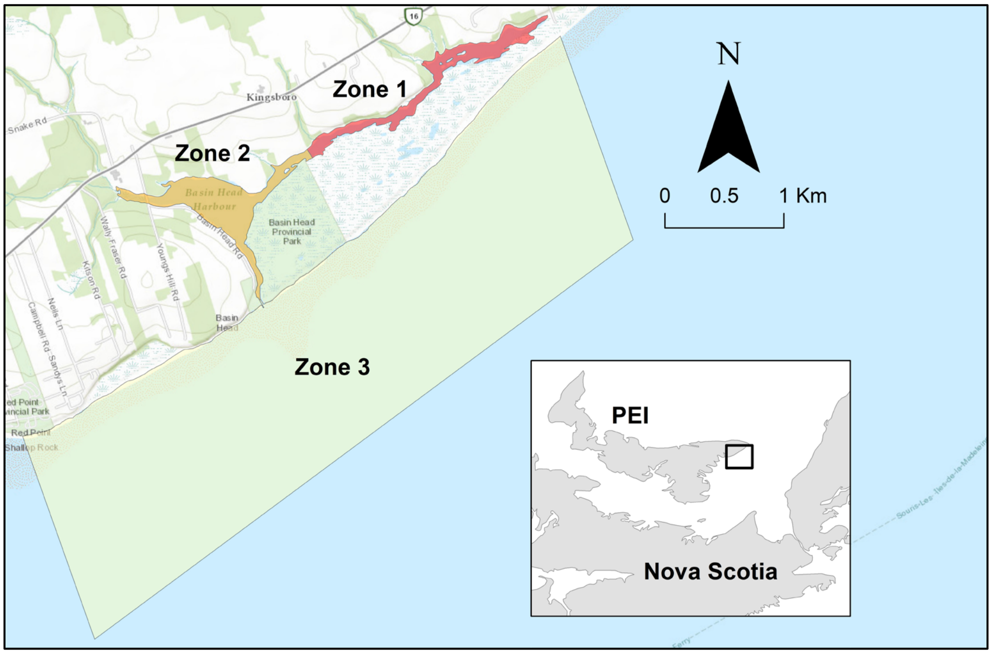

Figure 5 | The Boundaries of the Basin Head MPA. Zone 1 is the most protected part of the MPA and Zone 3 is the least protected. Inset shows the location of the MPA in relation to Prince Edward Island (PEI) and Nova Scotia. Source: Government of Canada (2020g).

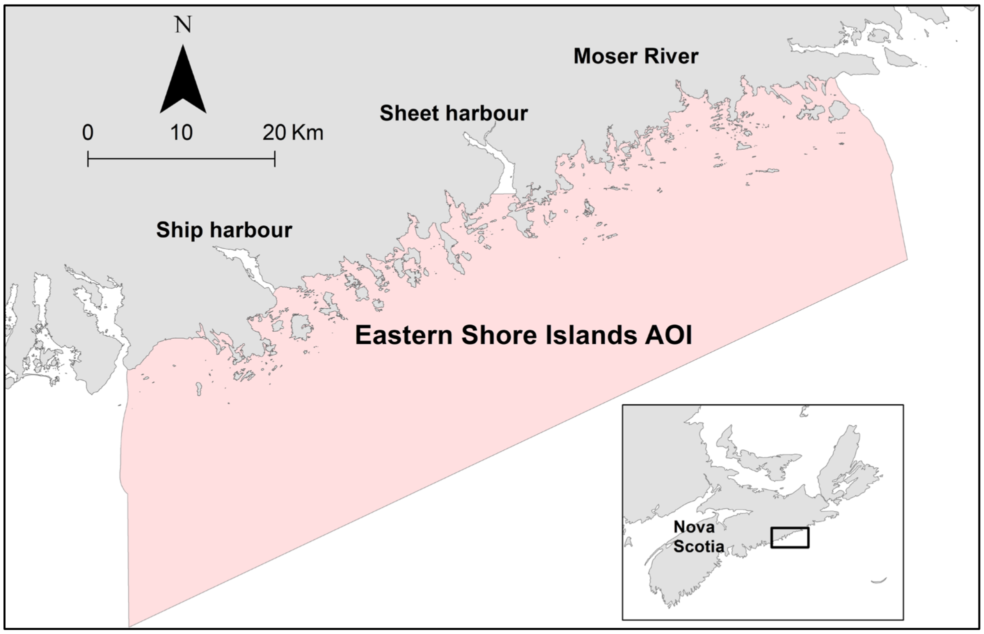

Figure 6 | The boundaries of the Eastern Shore Islands AOI located along the eastern shore of Nova Scotia. This area is under review for MPA designation, and if approved, would include the protection of eelgrass as one of its management objectives. Source: DFO.

4.3. Other spatial management areas

In addition to MPAs, other spatial management areas are likely to encompass eelgrass meadows in Atlantic Canada. For example, there are eelgrass beds in the Little Port Joli Estuary, located within Kejimkujik National Park Seaside Adjunct (Government of Canada 2020f). However, disturbance from invasive European green crabs (*Carcinus maenas*; see Section 5.2) have caused these eelgrass beds to decline by 98 % (Parks Canada 2016). Kouchibouguac National Park in New Brunswick also encompasses extensive eelgrass beds (Joseph et al. 2006). Lastly, several National Wildlife Areas (e.g. Boot Island) in Nova Scotia are likely to encompass eelgrass (Jeffrey Barrell, DFO, pers. comm. 16th July 2020) and some Migratory Bird Sanctuaries (e.g. Port Joli, Port l'Hebert, and Sable River) were designated partly because migratory geese feed on eelgrass in these areas (Melisa Wong, DFO, pers. comm, February 2021). Migratory Bird Sanctuaries and National Wildlife Areas are managed by the Canadian Wildlife Service (CWS), part of Environment Climate Change Canada (ECCC). For a full list of these sites, see:

- www.canada.ca/en/environment-climate-change/services/national-wildlife-areas.html and;
- www.canada.ca/en/environment-climate-change/services/migratory-bird-sanctuaries.html

4.4. Species at Risk Act (SARA)

The federal *Species at Risk Act* (SARA; S.C., 2002) aims to:

- Prevent the extinction of wildlife in Canada;
- Help the recovery of species that are 'Threatened', 'Endangered' or 'Extirpated' (i.e. a species that no longer exists in the wild in Canada but does elsewhere); and
- Prevent species of 'Special Concern' from becoming Threatened or Endangered (Government of Canada 2020e).

Under SARA, any actions that could harm or harass species on Schedule 1 of the List of Wildlife Species at Risk are legally prohibited within Canada (Government of Canada 2020h). The Committee on the Status of Endangered Wildlife in Canada (COSEWIC) are an independent panel of experts and are responsible for assessing the status of wildlife species and, if necessary, for recommending classifications that impose their legal protection. COSEWIC are also responsible for identifying 'critical habitats' that are key to the conservation of Schedule 1 species, and for developing plans and strategies (e.g. voluntary actions, stewardship measures, and legal action if necessary) to prevent their harm and destruction.

At present, eelgrass is not a Schedule 1 species, nor is it considered to be 'Threatened', 'Endangered', or of 'Special Concern', nor is it a 'Critical Habitat' of any other Schedule 1 species. However, there are several COSEWIC-assessed species that are closely associated with eelgrass meadows in Atlantic Canada, which are reviewed in the following sections:

4.4.1. Barrow's goldeneye

The eastern population of Barrow's goldeneye is designated 'Special Concern' on Schedule 1 of the List of Wildlife Species at Risk (Government of Canada 2020c). A very small proportion (approximately 400 individuals) of this population overwinter in Maine and Atlantic Canada. Barrow's goldeneye have been reported to use estuaries containing eelgrass in Atlantic Canada (Hanson 2004a).

4.4.2. American eel

The American eel (*Anguilla rostrata*) is considered to be 'Threatened' by COSEWIC and consultations are currently underway to determine if it should be listed on Schedule 1 of the List of Wildlife Species at Risk (Government of Canada 2020a). Evidence from Chesapeake Bay suggest American eels may inhabit eelgrass meadows (Orth and Heck 1980) and a DFO report states that eelgrass meadows in Atlantic Canada are "frequently used as habitat of eels in estuaries" (Chaput et al. 2013). However, there is currently no direct evidence that American eels are dependent on eelgrass meadows in Atlantic Canada.

4.4.3. Atlantic cod

Under SARA, Atlantic cod are divided into four discrete populations: the Maritimes populations (divided into the Laurentian South and the Laurentian North populations); the Newfoundland and Labrador population; and the Arctic population. Except for the Arctic population, all are considered to be 'Endangered' by COSEWIC but are not listed on Schedule 1 of the List of Wildlife Species at Risk (Government of Canada 2020b). As previously discussed (see Section 3.5), eelgrass beds provide important nursery habitats to juvenile cod in Newfoundland and Labrador, and to a lesser extent, in Nova Scotia (Melisa Wong, pers comm, 16th July 2020).

4.4.4. Atlantic salmon

In Nova Scotia, wild Atlantic salmon (*Salmo salar*) are divided into four discrete populations: Inner Bay of Fundy; Southern Upland; Eastern Cape Breton; and the Gulf of St Lawrence. Except for the Gulf of St Lawrence, all are considered to be 'Endangered' by COSEWIC but only the Inner Bay of Fundy population is listed on Schedule 1 of the List of Wildlife Species at Risk (Fisheries and Oceans Canada 2020c). There is no direct evidence that Atlantic salmon use eelgrass meadows in Nova Scotia, but they are thought to use eelgrass beds in the Gulf of St. Lawrence, and in Newfoundland and Labrador (Heike Lotze, Dalhousie University, pers. comm. 20th July 2020). The general lack of evidence is believed to be a result of their low abundance, which would reduce the probability of salmon appearing in dive and video surveys of eelgrass meadows. It is probable that their historic migration routes overlapped with eelgrass meadows in Nova Scotia (Heike Lotze, Dalhousie University, pers. comm. 20th July 2020).

4.4.5. Eelgrass limpet

The eelgrass limpet (*Lottia alveus*) is considered to be 'Extinct' by COSEWIC (Government of Canada 2020d). This mollusc fed exclusively on the epithelial tissues (i.e. the outer-most layer) of eelgrass and was once distributed throughout the Northwest Atlantic coast. The large outbreak of wasting disease during the 1930's caused an unprecedented decline in eelgrass cover (see Section 5.1), and led to the extinction of this species.

4.5. The Fisheries Act and HADD provisions

Section 35 of the *Fisheries Act* (R.S.C. 1985) legally prohibits the "harmful alteration, disruption or destruction" (HADD) of eelgrass and other fish habitat. This means that if a proponent damages fish habitat, they could be prosecuted for breaching federal law. However, exceptions are granted to works, undertakings or activities that may cause a HADD if they are 'prescribed' or belong to a 'prescribed class', or if authorization is granted from a federal authority following an environmental assessment (Fisheries and Oceans Canada 2020f). To gain authorization, the proponent must demonstrate how they intend to minimize habitat impacts, and the authorization will usually impose responsibilities for mitigation, habitat restoration or offsetting, and monitoring. This legislation has important implications for a variety of activities, including aquaculture (see Section 8 on aquaculture and eelgrass management).

The concept of HADD has a long and complex history. The *Fisheries Act* first came into force in 1868 and outlined a series of legal measures aimed at conserving and protecting fish and their associated habitats (Fisheries and Oceans Canada 1995). HADD was then introduced to the *Fisheries Act* in 1977 to strengthen protection of fish and their habitats. In 2012, the *Fisheries Act* was revised and HADD was removed and replaced with a prohibition on works, undertakings or activities that result in "serious harm to fish that are part of a commercial, recreational or Aboriginal fishery", commonly referred to as CRA. Scientists, environmental groups, and other stakeholders expressed concerns that these amendments could lessen the protection afforded to wildlife as the change from "harmful alteration, disruption or destruction" to "serious harm" might permit greater levels of environmental damage, and that only species targeted by fisheries were protected, rather than all species (Wilt 2018). In response, the Federal Government led a review in 2016 and HADD was reintroduced to the *Fisheries Act* in 2019.

To help assist federal and provincial regulators determine what may constitute a HADD to eelgrass, DFO held a science peer-review meeting in 2011 (Fisheries and Oceans Canada 2012) and proposed the following interpretations:

- **No effect:** Eelgrass bed integrity is not compromised. No observable changes in eelgrass structure, within natural variation. Fish habitat function is unaffected or improved.
- **Disruption:** Eelgrass beds will recover their structure and integrity within one year. Patchiness is increased only to the point that recolonization of bare areas, increased density, or return to original meadow size can occur within one year.
- **Harmful alteration:** After a year or more, eelgrass beds will only recover part of their structure and integrity. Patchiness is permanent (relative size of corridors to patches), shoot density will remain low, meadow size is reduced for more than one year.
- **Destruction:** Eelgrass meadow will not survive beyond the season and will not recover without intervention.

The reviewers also identified five key stressors that could cause a HADD to eelgrass (see Section 5 for greater detail on stressors to eelgrass):

1. **Sedimentation:** which may cause burial of eelgrass.
2. **Light limitation:** which may reduce eelgrass growth and lead to mortality.
3. **Nutrient loading:** which may lead to anoxia, nitrogen toxicity, and sulphide accumulation.
4. **Water flow:** Which controls sediment erosion and eelgrass distribution.
5. **Physical damage:** Which can cause immediate rhizome / shoot damage.

Thresholds were proposed for some of these stressors but the peer-reviewers acknowledged that these are largely unrealistic because: (1) eelgrass stressors rarely act in isolation and little is known about their interactive and cumulative effects (see Section 5.7); (2) stressors act on eelgrass against a background of high natural variability (see Section 2); (3) eelgrass recovery is highly variable and depends on the strength of the seed bank and connectivity to nearby patches (see Section 2); and (4) that most scientific evidence for these thresholds come from highly controlled, short-term laboratory studies that do not reflect long-term eelgrass dynamics under real world conditions (see Section 5). Consequently, this remains an active area of research and policy development (Jeffrey Barrell, DFO, pers. comm. 21st July 2020).

5. Eelgrass declines in Nova Scotia

Some eelgrass beds in Atlantic Canada have declined in recent years (Garbary and Munro 2004, Malyshev and Quijón 2011, Murphy et al. 2020). Anecdotal reports and scientific observations suggest that eelgrass meadows occupied most suitable intertidal mud flat areas in Nova Scotia until the late 1970's (Sharp and Semple 2004). However, by the early 1990's to 2000's, many of these areas had little to no eelgrass cover (Seymour et al. 2002, Chapman and Smith 2004, Sharp and Semple 2004, Garbary et al. 2014). The declines within Antigonish estuary (see Section 5.2) are perhaps the most dramatic and well documented, as eelgrass meadows in this area experienced a 95 % reduction in biomass between 2000 – 2001 (Seymour et al. 2002, Garbary et al. 2014). Similar declines have been reported in Petpeswick, which underwent a 96 % reduction in eelgrass cover between 1992 – 2002, while eelgrass beds in Cole Harbour declined by just 49 % during the same time period (Chapman and Smith 2004). On a longer time-scale, eelgrass beds in Lobster Bay underwent a 30 – 44 % reduction in eelgrass cover between 1978 – 2000 (Sharp and Semple 2004).

To discuss the possible drivers underlying these declines, and the issues surrounding eelgrass mapping and monitoring, a technical workshop (attended by over 40 experts) was held in New Brunswick in 2003 (Hanson 2004b). It was concluded that no single casual factor was likely to be responsible. Rather, a multitude of interacting factors were likely causing eelgrass declines in Nova Scotia, as detailed in the following sections.

5.1. Wasting disease

In the early 1930's, eelgrass populations across the Atlantic coasts of North America and Europe were decimated by an outbreak of 'wasting disease' (Muehlstein 1989). This disease is caused by an infectious slime mold (*Labyrinthula zostera*) that spreads via direct leaf-to-leaf contact and causes eelgrass shoots to develop black-brown dots and streaks, eventually leading to their mortality. It is thought that the outbreak in North America started in Virginia in 1930, which spread northwards to Eastern Canada. By 1931, more than 90 % of eelgrass beds had disappeared along the Northwest Atlantic coast, rising to 99 % the following year. Although recovery was relatively slow, many eelgrass beds had re-established by the early 1950's. A recurrence of the disease was documented across New England in 1984, and led to significant declines of up to 80 % in some populations (Short et al. 1988).

Wasting disease continues to affect eelgrass beds in North America and Europe with variable degrees of loss (Garbary and Munro 2004, Moore and Short 2006) and can be detected at low levels in eelgrass beds throughout Atlantic Canada (Jeffrey Barrell, DFO, pers. comm. 29th September 2020). The general consensus is that the disease is not responsible for the present-day declines in eelgrass in Atlantic Canada (Garbary and Munro 2004, Garbary et al. 2014), although it could potentially be an important factor in populations subject to multiple stressors (see Section 5.7).

5.2. Invasive European green crab

The European green crab (Figure 7) is native to coastal waters in the Northeast Atlantic, and the Baltic and North Seas, with its distribution traditionally ranging from Norway to Northwest Africa (Grosholz and Ruiz 1996). However, new populations have established along the Pacific and Atlantic coasts of North America, as well as in South Africa, Australia, and New Zealand. Consequently, the European green crab is considered to be one of the world's most invasive species (Fisheries and Oceans Canada 2020e). It was first detected in North America during the early 1800s in New England (Carlton and Cohen 2003, Matheson et al. 2016). The population then expanded into the Bay of Fundy during the 1950's. By 2007, sightings of European green crab had been reported all across Atlantic Canada, from Nova Scotia to the southern shore of Newfoundland (Fisheries and Oceans Canada 2020e).

Figure 7 | A European green crab (*Carcinus maenas*) inhabiting an eelgrass meadow in Kejimkujik National Park Seaside. Source: Parks Canada.

The European green crab is a highly aggressive and voracious predator that can outcompete native species for food. In Atlantic Canada, it can prey on a variety of intertidal organisms including oysters, mussels, clams, and native crab species (Grosholz and Ruiz 1996, Klassen and Locke 2007, Fisheries and Oceans Canada 2020e). It can also damage eelgrass by feeding on the base of their shoots (Malyshev and Quijón 2011) and by uprooting their roots and rhizomes while digging for clams and other invertebrates buried within the sediment (Seymour et al. 2002, Garbary and Munro 2004). Evidence from a field enclosure experiment in Tracadie Harbour, Nova Scotia, suggested that European green crabs within the harbour can remove up to 87,000 eelgrass shoots (~ 890 kg) per day (Garbary and Munro 2004).

In Nova Scotia, damage from European green crabs is thought to be the primary mechanism responsible for a 95 % reduction in eelgrass cover in Antigonish between 2000 – 2001, as European green crabs had reached an abundance of 385,000 individuals per km^2 during this time (Campbell 2001, Seymour et al. 2002). This notion was reinforced after surveys in 2013 observed European green crabs had reduced to < 1 individual per km^2, and eelgrass cover had recovered to 60 % of its pre-2000 values (Garbary et al. 2014). European green crabs are also thought to be partly responsible for a 98 % reduction in eelgrass cover in Kejimkujik National Park Seaside between 1987 – 2010 (Parks Canada 2016). Lastly, a survey of 13 estuaries in New Brunswick, PEI and Nova Scotia between 2001 – 2002, found eelgrass biomass was generally lower in estuaries invaded by European green crab, compared to uninvaded ones (Locke and Hanson 2004).

5.3. Reduced light availability

Seagrasses, like all plants, require light to photosynthesize sugars and other carbohydrates necessary for respiration and growth. Consequently, light availability is one of the most important factors controlling seagrass growth (Dennison and Alberte 1985, Duarte et al. 2006, Schmidt et al. 2012). There are many natural and human sources of disturbance that can reduce light availability and impact seagrasses, including sedimentation and sediment resuspension from storms, river discharge, coastal construction, moorings and dredging (see Section 5.6). Shading from marinas and aquaculture infrastructure (see Section 7) can also reduce the amount of light available to seagrass, as can eutrophication, which can cause excessive phytoplankton and epiphyte growth (see Section 5.4). Conversely, there is some evidence that suspended oyster aquaculture can increase eelgrass growth by improving water clarity, reducing epiphyte loads, and providing more nutrients to eelgrass (see Section 7.4.1)

Burke et al. (1996) conducted several experimental field manipulations in Virginia, USA, and reported that shading eelgrass for three weeks led to reductions of 40 – 51 % in tissue sugar concentration, 34 % in leaf biomass, 27 % in shoot density, and 23 % in root and rhizome biomass. Similar field manipulations have been conducted in Nova Scotia and have yielded similar results (Wong et al. 2020). Such negative responses tend to get stronger with longer durations of light reduction (Ralph et al. 2006). For example, a laboratory study conducted by Bertelli and Unsworth (2018) demonstrated that reducing light levels below 20 μmol photons $m^{-2}\,s^{-1}$ resulted in significant reductions in eelgrass growth and photosynthetic performance after 7 days, a 41 % reduction in leaf size after 29 days, and shoot mortality within 4 – 6 weeks (Bertelli and Unsworth 2018). Burial under sediments can also affect seagrass by reducing the area of the plant available for photosynthesis. For instance, a field manipulation study by Mills and Fonseca (2003) showed that eelgrass buried up to 25 % of their height for 24 – 28 days resulted in a 75 % mortality rate, which increased to 100 % between burial depths of 50 – 75 % (Mills and Fonseca 2003). Thus, it was concluded that eelgrass has a particularly low threshold for tolerating burial.

5.4. Nutrient enrichment and eutrophication

Nutrient concentrations can become elevated in coastal waters, rivers and estuaries due to the release of effluents from agriculture, aquaculture (see Section 7), industrialization, urbanization, wastewater treatment plants, and other human activities (Nixon 1995, Smith 2003).

Elevated nutrient concentrations can reduce light levels by promoting the growth of: (1) phytoplankton, which can reduce water clarity; (2) benthic macroalgae, which can compete with seagrasses for light and space; and (3) epiphytic algae and other organisms which grow on the blades of seagrass, obstructing them from light (Williams and Ruckelshaus 1993, Short et al. 1995, Hauxwell et al. 2001, McGlathery 2001, Hauxwell et al. 2003). A survey of 12 estuaries in PEI and New Brunswick between 2007 – 2008 showed that those with elevated nutrient levels supported almost double the biomass of phytoplankton, 40 times more epiphytic algae, and 670 times more opportunistic green macroalgae (Schmidt et al. 2012). Due to restricted light availability, the eelgrass growing in these nutrient enriched estuaries exhibited significantly lower shoot density, as well as lower above and below ground biomass (Schmidt et al. 2017).

As well as restricting sunlight, high biomasses of phytoplankton and macroalgae can cause greater quantities of detritus and organic matter to settle and decompose on the seafloor. In oxygenated environments, bacteria decompose this organic matter through aerobic respiration, consuming oxygen in the process. However, excessive quantities of organic matter can cause bacteria to partially ('hypoxia') or fully ('anoxia') deplete oxygen, prompting bacteria to switch to anaerobic respiration, which can cause hydrogen sulphide (H_2S) and other sulphide compounds to build-up within the sediment (Bricker et al. 2007, Schmidt et al. 2012, Benson et al. 2013). Eelgrass is relatively tolerant to anoxia compared with other seagrass species, but low oxygen levels can reduce its metabolism and growth (Pregnall et al. 1984, Smith et al. 1988). Sulphides are potentially a bigger threat as they are toxic to seagrasses and have been shown to significantly affect eelgrass photosynthesis, metabolism, leaf size, and shoot height, which can lead to their mortality (Carlson et al. 1994, Goodman et al. 1995, Terrados et al. 1999, Pedersen et al. 2004). For example, Dooley et al. (2013) observed that eelgrass seedlings were consistently killed when exposed to water H_2S concentrations above 680 µM. The degree to which sulphides impact eelgrass is strongly linked to oxygen concentrations both within the water column and sediment, as eelgrass can resist sulphides from entering their tissues provided their roots and rhizomes are supplied with sufficient levels of oxygen (Pedersen et al. 2004).

Another potential impact of nutrient enrichment is nitrogen toxicity. Effluents from human activities can release nitrate (NO_3^-) and ammonium (NH_4^+) into coastal waters, which can be toxic to seagrasses when present in high concentrations. Burkholder et al. (1992) maintained eelgrass in elevated water NO_3^- concentrations of approximately 200 ~ 300 µM for 8 weeks, and found that it caused their shoots to crumble, which eventually led to their mortality (Moore and Wetzel 2000). Likewise, van Katwijk et al. (1997) observed that water NH_4^+ concentrations of 25 µM adversely affected eelgrass, and that concentrations of 125 µM led to their mortality within 2 – 5 weeks. Interestingly, seagrasses are more tolerant to high nitrogen concentrations within the sediment than in the water column. For instance, Peralta et al. (2003) demonstrated that eelgrass could tolerate sediment NH_4^+ concentrations up to 30 mM, which is 1200 times higher than what they can tolerate in the water (van Katwijk et al. 1997). Nitrogen toxicity also depends on sediment type, as eelgrass has been shown to be less sensitive to NH_4^+ when growing in muddy sediments compared to sand (van Katwijk et al. 1997).

Overall, the process of nutrient enrichment leading to hypoxia and increased algal biomass, known as 'eutrophication', is considered to be one of the most important drivers underlying the loss of seagrass worldwide (Kenworthy et al. 2006, Walker et al. 2006). However, the effects of nutrient enrichment and eutrophication on seagrasses are highly complex, and can be strongly influenced by a range of other factors including sediment composition, light availability, temperature, oxygen concentration and sediment redox potential (McGlathery 2001, Walker et al. 2006).

5.5. Warming temperatures and increasing storms

Ocean temperatures have displayed general warming trends over the last three decades in the Bay of Fundy, Scotian Shelf, Cabot Strait, Northumberland Strait, and Gulf of St Lawrence. Since records first began in

1985, three of the five warmest years have occurred in 2012, 2014, and 2015 (Herbert and Pettipas 2016, Bernier et al. 2019). However, some parts of the Bay of Fundy and Halifax Harbour have exhibited a general decrease in temperature, or no significant change in temperature (Herbert and Pettipas 2016). Nonetheless, ocean temperatures in Canada are projected to continue increasing over the 21st century, and the waters in Southern Atlantic Canada (Figure 1) are expected to warm faster than the rest of the country (Greenan et al. 2019, Lavoie et al. 2020). Overall, predictions under a high emissions scenario suggest summer sea surface temperatures may increase by 4 °C by 2050 in Atlantic Canada (Greenan et al. 2019).

Eelgrass is widely distributed across the sub-Arctic, temperate and sub-tropical regions of the Pacific and Atlantic Oceans, indicating that it can tolerate a wide a range of temperatures. However, the rate at which ocean temperatures are rising, coupled with an increasing occurrence of unusually warm and long summer temperatures (Greenan et al. 2019), could pose a threat to eelgrass meadows in Atlantic Canada. A laboratory study by Nejrup and Pedersen (2008) found that eelgrass collected from Danish estuaries experienced a 12-fold increase in shoot mortality when exposed to temperatures of 25 – 30 °C compared to 10 – 20 °C. Temperature studies are often confounded by oxygen concentrations, since warmer waters inherently contain less oxygen. Hammer et al. (2018) avoided this issue by exposing eelgrass collected from Virginia, USA, to elevated temperatures, while maintaining oxygen saturation at 100 %. Their study determined that temperatures of 26 °C and 30 °C negatively affected leaf growth, leaf formation, rhizome growth, root formation, and survival compared to eelgrass incubated at 22 °C. These negative relationships could explain why Reusch et al. (2005) lost half of their experimental eelgrass plots in the Baltic Sea after a summer heatwave caused water temperatures to exceed 25 °C. These effects may also partly explain why Wong et al. (2013) observed lower biomass, production and growth of Nova Scotian eelgrass beds in Kejimkujik compared to Port Joli and Port L'Hebert, as water temperature temperatures were higher in Kejimkujik. Paradoxically, increases in eelgrass cover have been reported for most of Newfoundland, possibly due to warmer temperatures reducing scouring by sea ice (Bernier et al. 2019).

Sea levels, and the frequency of flooding and storm events, are also projected to increase with rising temperatures in Atlantic Canada (reviewed in Atkinson et al. 2016, Lemmen et al. 2016, Rapaport et al. 2017). These changes could increase the susceptibility of eelgrass to erosion, dislodgement, sediment burial, and turbidity and salinity changes, which could lead to a general alteration in habitat suitability of existing eelgrass locations (reviewed in Perry et al. 2019).

5.6. Mechanical damage

Seagrasses are generally restricted to shallow areas that are sheltered from large waves and strong winds. These sheltered locations are also attractive areas for the anchoring and mooring of boats. Tides and winds cause boats to rotate around a central anchor point, causing their mooring chains to drag across the seafloor over a fixed radius (Hastings et al. 1995). This can cause repeated physical disturbance to any underlying seagrass (Figure 8) by tearing shoots and uprooting rhizomes (Bourque et al. 2015, Glasby and West 2018). A study in the south-west of England documented that each individual boat mooring resulted in the loss of over 120 m^2 of eelgrass (Unsworth et al. 2017). Dragging of anchors and mooring chains can also resuspend sediments and increase the risk of burial (Unsworth et al. 2017, Glasby and West 2018). There are many 'seagrass friendly' mooring designs available to boaters that reduce damage to seagrass, and several scientists have argued that their use should be legally imposed by legislation (e.g. Demers et al. 2013, Luff et al. 2019).

Figure 8 | Photo of Lake Macquarie in New South Wales, Australia. Each boat mooring has removed a clear radius of seagrass from the seabed. Source: Dr Tim Glasby.

Motorboats can also cause 'propeller scars' by removing and damaging seagrass leaves, shoots and rhizomes (Zieman 1976, Dawes et al. 1997). Such scars are extensive across Atlantic Canada (Jeffrey Barrell, DFO, pers. comm. 29th September 2020). Finally, coastal construction, dredging, and fishing activities (e.g. scallop dredging and oyster tonging) can damage seagrass shoots and reduce their growth through sediment resuspension (Fonseca et al. 1984, Short and Wyllie-Echeverria 1996, Erftemeijer and Robin Lewis 2006, Nordlund et al. 2018).

5.7. Multiple stressors and their interactive effects

The previous sections of this report describe how seagrasses can be affected by a wide range of natural and anthropogenic stressors. However, it is very rare for stressors to occur in isolation. For instance, Murphy et al. (2019) developed a metric to assess the cumulative impact of multiple human activities on seagrass. By applying this metric to 180 eelgrass beds in Atlantic Canada, they found eelgrass existed across a wide spectrum of human impacts including nutrient enrichment, species invasions, fishing, aquaculture, and coastal construction. A growing number of studies show that multiple stressors can interact, and the effects of one can cause seagrass to become more sensitive to another (Blake and Duffy 2012, Brown et al. 2014, Stockbridge et al. 2020, Vieira et al. 2020, Krumhansl et al. 2021). Conversely, some stressors have been shown to have no interactive effects, while others can reduce the sensitivity of seagrass to other stressors (Blake and Duffy 2010, York et al. 2013, Mvungi and Pillay 2019). Consequently, it is difficult to isolate or predict the effects of a single stressor on seagrass populations in a field-based setting, especially considering their spatial dynamics and annual and seasonal fluctuations (see Section 2).

5.8. Aquaculture

Aquaculture describes the culture of aquatic plants (e.g. seaweeds and algae) and animals (e.g. finfish and shellfish) grown in the sea, areas of freshwater, or in tanks on land. As the majority of aquaculture in Atlantic Canada (see Section 6.1) occurs within coastal waters, it has potential to coincide and interact with eelgrass meadows. The remainder of this report focuses on these potential interactions.

6. Aquaculture in Atlantic Canada

The aquaculture industry in Atlantic Canada has exhibited significant growth since the mid-1980's (Figure 9). New Brunswick is the largest aquaculture producer, followed by PEI, Newfoundland and Labrador, and finally by Nova Scotia. Finfish represent over 80 % of all aquaculture production in the Maritimes, except in PEI where production is almost exclusively focused on shellfish.

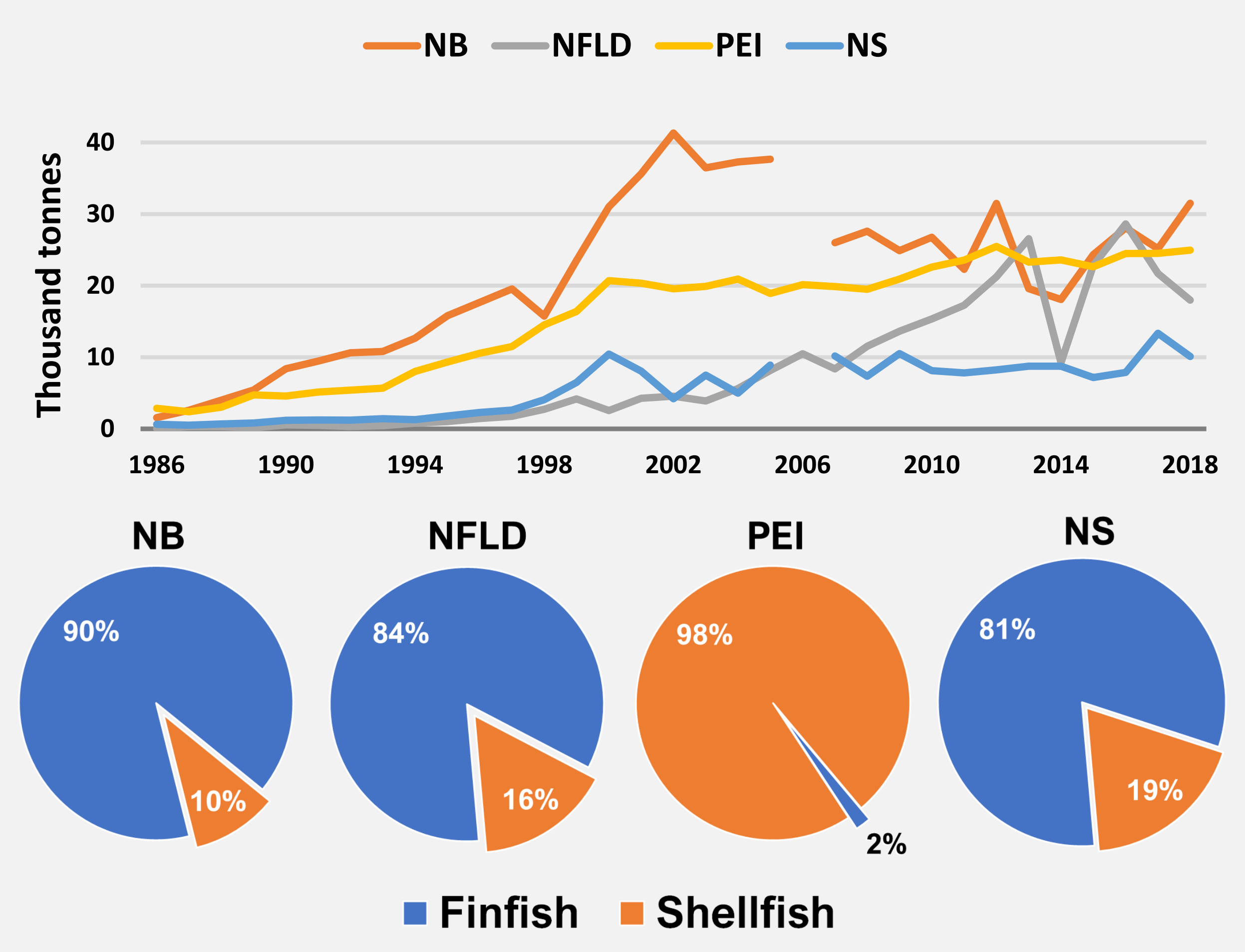

Figure 9 | Annual levels of aquaculture production in the Maritime provinces (top) and the proportion of production in 2018 dedicated to finfish and shellfish (bottom). NB = New Brunswick, NFLD = Newfoundland and Labrador, PEI = Prince Edward Island, and NS = Nova Scotia. Source: Fisheries and Oceans Canada (2020b). Gaps represent years with no available data.

6.1. Aquaculture in Nova Scotia

The aquaculture industry in Nova Scotia has seen substantial growth since the early 1990's, with production increasing five-fold since 1995 (Figure 10). This growth is mostly due to the expansion of the Atlantic salmon industry, and to a much lesser extent, for steelhead / rainbow trout (*Oncorhynchus mykiss*). Consequently, finfish aquaculture now represents 81 % of all aquaculture production by weight (8,201 tonnes in 2019) and 93 % by value ($69.5 million in 2019).

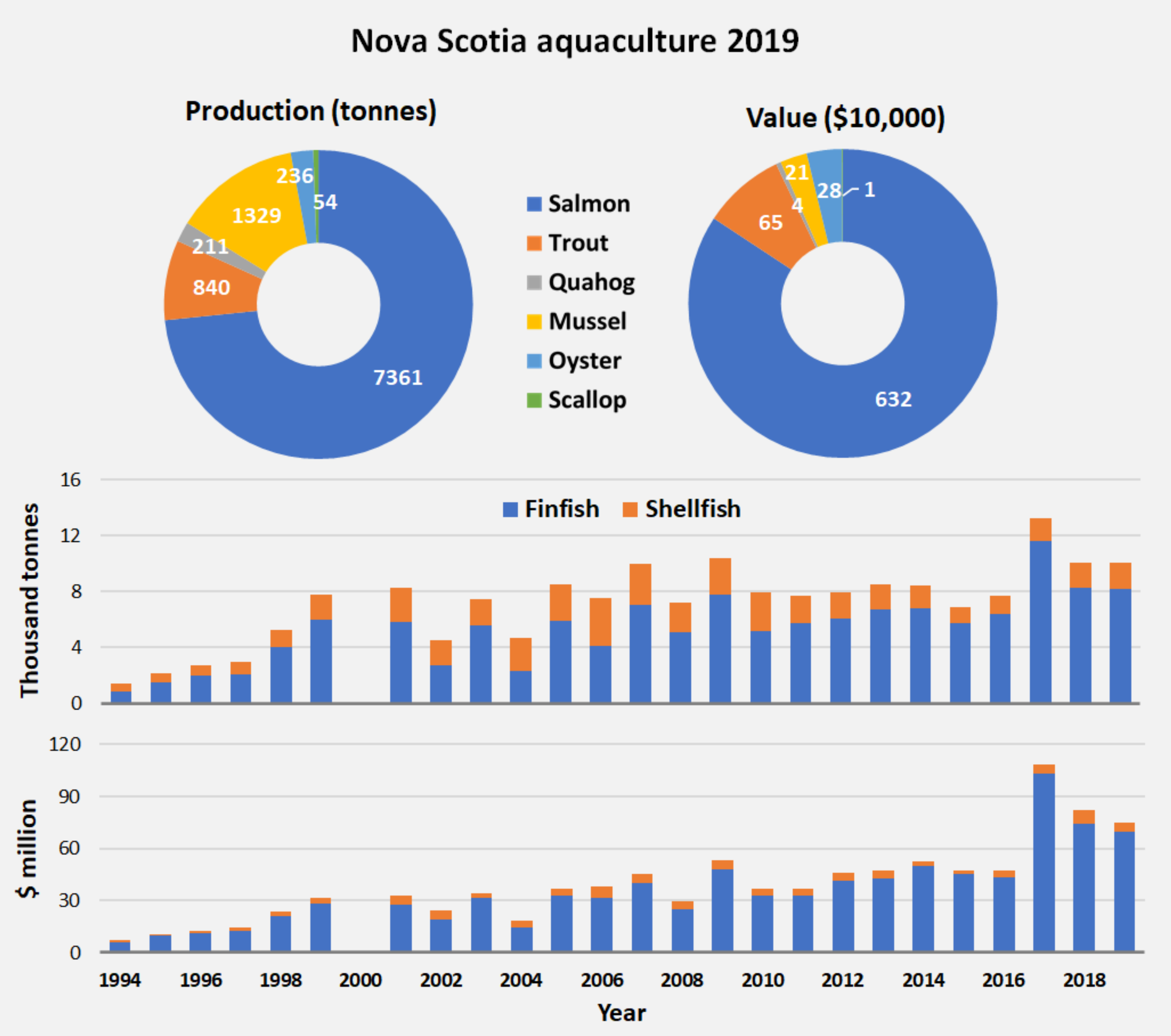

Figure 10 | Aquaculture production levels and value in Nova Scotia for 2019 divided by species (top), and over time (bottom) divided by finfish and shellfish. Salmon = Atlantic salmon, trout = rainbow and brook trout, mussel, = blue mussel, oyster = American oyster, scallop = sea scallop. Source: Nova Scotia Department of Fisheries and Aquaculture (2020c). Gaps represent years with no available data.

Although finfish aquaculture dominates production and value in Nova Scotia, there are currently many more marine aquaculture leases issued for the production of shellfish (169 leases) than finfish (35 leases) (Nova Scotia Department of Fisheries and Aquaculture 2020a). Blue mussels (*Mytilus edulis*) make-up the majority of shellfish production in Nova Scotia, representing 13 % of all aquaculture production by weight (1,329 tonnes in 2019) and 3 % by value ($2.1 million in 2019). Production of American oyster (*Crassotrea virginica*) generates comparatively more value ($2.7 million in 2019) than blue mussel, despite production being substantially lower (236 tonnes in 2019). A small number of shellfish growers in Nova Scotia produce clams (quahog, *Arctica islandica*, and soft-shell clams, *Mya arenaria*), and scallops (bay scallop, *Argopecten irradians*, and sea scallop, *Placopecten magellanicus*). There are also initiatives underway to facilitate further growth of the shellfish aquaculture industry such as the proposed Aquaculture Development Area (ADA) in Lobster Bay. This is currently under assessment by Nova Scotia Department of Fisheries and Aquaculture (NSDFA) and the Municipality of the District of Argyle (www.aquacultureargyle.com) and will focus primarily on shellfish and marine plants. ADA's aim to attract investment from growers to establish new operations within pre-defined areas that have already been assessed, through a public process, to be socially, environmentally, and economically suitable for aquaculture development (Matthew King, NSDFA, pers. comm. 30th July 2020).

In Nova Scotia, 76 % of all issued aquaculture leases are situated in waters less than 5 m deep, of which 96 % are within 800 m from the shore. These leases all produce shellfish as finfish production generally requires greater depths (see Section 6.2.1). Consequently, there is greater potential for shellfish aquaculture to directly overlap with suitable eelgrass habitat than finfish aquaculture (Figure 11).

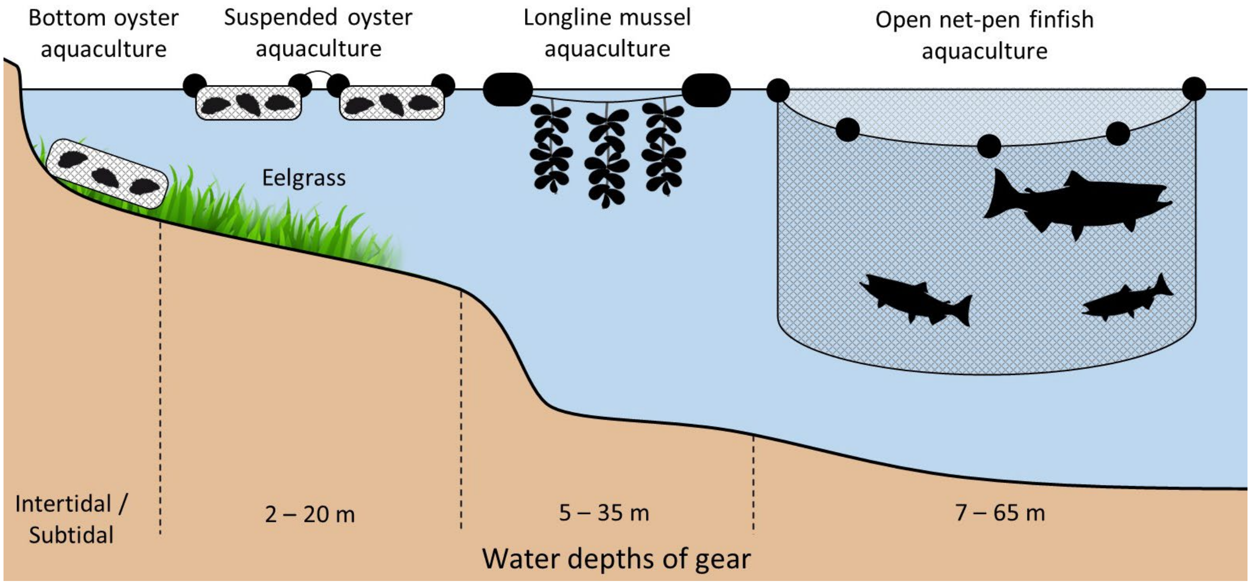

Figure 11 | Schematic diagram indicating the depth range of common aquaculture production methods in Nova Scotia. As depth increases, their chance of overlapping with eelgrass habitat diminishes. Diagram is not to scale.

6.2. Overview of production methods

6.2.1. Finfish aquaculture

In Nova Scotia, finfish farm leases are generally situated in coastal waters 100 – 1700 m from the shore in depths of 7 – 65 m (data range provided by Nova Scotia Department of Fisheries and Aquaculture - NSDFA). Farms typically comprise of 2 – 20 circular net-pens, measuring 20 – 40 m in diameter, which extend approximately 8 – 10 m downwards into the water (Nathaniel Feindel, NSDFA, pers. comm. 26th August

2020). Other site infrastructure include large moorings which anchor the pens to the seafloor, inner and outer marker buoys, and large facilities have feed barges which deliver feed pellets to the pens via a series of surface pipes and blowers (Figure 12). Salmon production is typically an 18 month cycle which begins with the stocking of hatchery-reared smolts (body mass ~110 g) in the spring (Reviewed in Chang 1998). These are then harvested as full-size adults (body mass ~6 kg) during the winter of the following year. Trout production follows a similar process and timeline.

Figure 12 | A typical finfish farm in Nova Scotia comprising of two rows of 7 pens, a feed barge, feed pipes, and inner and outer marker buoys. Source: Kevin Schyf.

6.2.2. Mussel aquaculture

Mussel farms in Nova Scotia are generally situated in coastal waters 150 – 1200 m from the shore in depths of 5 – 35 m (Nova Scotia Department of Fisheries and Aquaculture 2020a). Most mussel farms use a suspended longline system (Figure 13), where multiple longlines are suspended in the water by surface buoys (Reviewed in Scarratt 2000, Fisheries and Oceans Canada 2015, Clements and Comeau 2019). Longlines typically measure between 100 – 200 m in length and can be sunk to deeper waters to avoid winter sea ice. Production usually begins in May – June, when wild mussel seed are collected on lengths of rope, or mesh, hanging down from the longlines. The seed are then allowed to grow until late October – November until they reach around 12 – 20 mm long. Following this, the seed are harvested and placed inside polyethylene 'sleeves' or 'socks' measuring up to 2 m in length which are hung from the longlines. It then takes the mussels around 15 – 24 months to reach a marketable size.

Figure 13 | A boat hauls up a series of mussel socks attached to a single longline. The longline is suspended in the water by multiple surface buoys. Source: Aaron Ramsay.

6.2.3. Oyster aquaculture

In Nova Scotia, oyster farms are generally situated in coastal waters ranging in depth from the intertidal zone to 20 m, and located up to 3 km from shore (Nova Scotia Department of Fisheries and Aquaculture 2020a). Several different production methods are used in Atlantic Canada but most growers use suspended bag or suspended cage systems. This involves growing oysters inside mesh bags or cages which are suspended at, or just below, the surface from a series of longlines. Rope culture is an alternative form of suspension oyster aquaculture and involves directly attaching oysters to lengths of rope hanging from the longlines. Like mussel farming, oyster longlines can be held afloat by surface buoys, or the bags / cages themselves may be equipped with floats (Figure 14). To avoid sea ice and storms, bags / cages can be submerged to rest on the seafloor within the lease site, or within a separate lease site dedicated to overwintering. Not all oyster growers in Nova Scotia sink their gear during the winter but are generally prepared to do so if water temperatures drop below 2°C (Scott Samson, Louisberg Seafoods, pers. comm, March 2021). Some growers in Nova Scotia use a more traditional 'on-bottom' production method where oyster bags or cages rest directly on the sediment within the shallow subtidal zone, and can be exposed during low tide (Nathaniel Feindel, NSDFA, pers. comm. 27th August 2020). In some cases, oysters are seeded directly onto the seabed without use of gear (Jeffrey Barrell, DFO, pers. comm. 29th September 2020).

Figure 14 | An oyster farm using floating cages. The cages can be rotated every few weeks to reduce biofouling and submerged to avoid winter sea ice. Source: Aaron Ramsay.

Most growers collect wild oyster seed by deploying spat collectors, or they purchase seed from other growers (Aaron Ramsay, Government of PEI, pers. comm. 31st August 2020). However, the collection of wild seed has become increasingly unreliable in Nova Scotia (Rod Beresford, Cape Breton University, pers. comm, 19th November 2020). While there are some oyster hatcheries in New Brunswick and Prince Edward Island, government regulations state that growers in the Bras d'Or lakes, Cape Breton, cannot import or export seed due to disease transmission risk (AAC 2012). Consequently, consistent and reliable access to seed is one of the biggest concerns for oyster growers in Nova Scotia (Mayer 2019). The grow-out period takes approximately 3 – 4 years before the oysters reach a marketable size (reviewed in Bastien-Daigle et al. 2007, Skinner et al. 2013).

6.2.4. Clam aquaculture

In Nova Scotia, clam aquaculture is conducted within the intertidal zone without any on-site infrastructure (Doug Bertram, pers. comm. 30th July 2020). Production begins by planting hatchery-reared seed into intertidal mud flat areas, known as 'seed beds'. The clams reach marketable size after approximately 20 months and are then harvested by hand using a rake, or 'hack' (Figure 15). Ideally, harvesting occurs at a rate that allows the seed bed to self-recruit and regenerate, without the need for more hatchery-reared seed. Consequently, clam aquaculture in Nova Scotia can be considered a type of 'enhanced fishery'.

Figure 15 | Two workers harvesting clams by hand from seed beds located on intertidal mud flats in Nova Scotia. Source: Doug Bertram.

6.2.5. Scallop aquaculture

Scallop farms in Nova Scotia are generally situated in coastal waters ranging between 5 – 35 m in depth, at a distance of 100 – 1500 m from shore (Nova Scotia Department of Fisheries and Aquaculture 2020a). Most farms produce sea scallops, however, some farms in the Gulf of St Lawrence produce bay scallops where water temperatures are suitably warmer. Growers of sea scallops generally purchase their seed from collectors based in Cape Breton (Duncan Bates, pers. comm. 30th July 2020). The seed are then suspended from longlines inside pearl or lantern nets, or via an 'ear hanging' system, where scallops are attached directly to the longlines (Figure 16). Due to strong tidal currents, growers in the Bay of Fundy use a different system where scallops are held in cages attached to frames anchored to the seafloor. Scallop seed take around two years to reach a marketable size.

Figure 16 | A Nova Scotia scallop farm utilizing a combination of ear hanging (left image) and lantern nets (right image). Source: Duncan Bates.

7. Aquaculture and the environment

7.1. The effects of finfish aquaculture on water and sediment biochemistry

The finfish aquaculture industry in North America and Europe primarily uses open net-pens situated in coastal waters. As open net-pens are designed to maximise water exchange, any resulting nutrient wastes are released into the surrounding water (Lawson 1995). These wastes can de divided into two categories; 'particulate' and 'dissolved'.

Particulate wastes derive from faeces and uneaten feed, and represent most of the carbon released from finfish farms (Islam 2005, Wang et al. 2012, Reid et al. 2013). Particulate wastes tend to settle quickly onto the seafloor and rarely disperse more than a few hundred metres (Brager et al. 2015, Price et al. 2015, Bannister et al. 2016, Filgueira et al. 2017). Consequently, they can accumulate under the pens and form a nutrient-enriched layer of organic matter overlying the sediment. Similar to the effects of eutrophication and nutrient enrichment (see Section 5.4), this organic matter can boost bacterial decomposition and lead to oxygen depletion and the build-up of sulphides within the sediment (Holmer et al. 2007, Pusceddu et al. 2007, Hargrave 2010, Price et al. 2015, Hamoutene et al. 2018). However, the quantity of particulate wastes produced by fish farms has been significantly reduced over the last three decades due to the development of more efficient feeds and feeding systems (Islam 2005, Sørensen 2012, Sprague et al. 2016).

Dissolved wastes are excreted by fish directly into the water column and represent the majority of nitrogen released from finfish farms (Norði et al. 2011, Wang et al. 2012). Up to 90 % of all the nitrogen excreted by marine finfish occurs as ammonia (NH_3), which quickly converts to ammonium (NH_4^+) at the pH of seawater (reviewed in Leung et al. 1999). Consequently, several studies have reported elevated NH_4^+ concentrations close to fish farms (Navarro et al. 2008, Sanderson et al. 2008, Jansen et al. 2018). Similar to eutrophication and nutrient enrichment (see Section 5.4), these elevated nitrogen concentrations can stimulate the growth of phytoplankton and macroalgae, thereby reducing oxygen levels and light availability (Cloern 2001, Robinson et al. 2005, Holmer et al. 2008b). However, a comprehensive review by Price et al. (2015) showed that most studies have found no direct evidence of fish farms increasing dissolved nitrogen concentrations of surrounding waters. This is partly because dissolved nitrogenous wastes can be quickly diluted and dispersed by tides and currents, rapidly assimilated by marine organisms (e.g. bacteria, phytoplankton, macroalgae and seagrass), and lost to the atmosphere through volatilization (Dalsgaard and Krause-Jensen 2006, Dailer et al. 2010). Hence, any localized increase in dissolved nitrogen is likely to be small, short-lived and difficult to detect (reviewed in Howarth et al. 2019). As a result, only finfish farms located in highly sheltered areas with low water exchange (i.e. low water turnover / high retention time) have been linked to eutrophication. For example, Pitta et al. (2005) investigated several fish farms in the Baltic Sea and only those located within small, shallow coastal bays (< 0.7 km^2) with low water turnover times (2 – 6 days) were found to increase phytoplankton and macroalgae growth.

7.2. Finfish aquaculture and seagrass interactions

7.2.1. Evidence from the Mediterranean Sea

Nearly all investigations into the impact of finfish aquaculture on seagrasses have been conducted in the Mediterranean Sea, and have examined the response of two species of Neptune grass (*Posidonia oceanica*, and to a lesser extent, *Cymodocea nodosa*) to finfish farms stocked with gilthead seabream (*Sparus aurata*) and European seabass (*Dicentrarchus labrax*). In general, these studies have reported decreasing seagrass cover with increasing proximity to finfish farms for distances up to 300 m, and the absence of seagrass directly under them (Table 2). These trends have been linked to increases in water and sediment nutrient concentrations, sediment organic matter, sedimentation, epiphyte loads, and increased grazing pressure from sea urchins (*Paracentrotus lividus* and *Arbacia lixula*) and other herbivores (reviewed in Holmer et al. 2008a, Cullain et al. 2018).

Table 2 | Overview of published responses of Neptune grass (*P. oceanica*, and *C. nodosa*) in close proximity to open net-pen finfish farms in the Mediterranean Sea.

Category	Parameter	Response	References
Seagrass physiology	Tissue carbohydrates	Decrease	Delgado et al. (1997), Ruiz et al. (2001)
	Photosynthesis	Decrease	Delgado et al. (1997), Cancemi et al. (2003)
Meadow structure	Above and below ground biomass	Decrease	Delgado et al. (1999), Apostolaki et al. (2009)
	Percentage cover	Decrease	Delgado et al. (1997), Ruiz et al. (2001), Holmer et al. (2008a)
	Shoot density	Decrease	Delgado et al. (1999), Pergent et al. (1999), Ruiz et al. (2001), Apostolaki et al. (2009), Rountos et al. (2012)
	Shoot mortality	Increase	Diaz-Almela et al. (2008), Holmer et al. (2008a)
Morphology	Leaf growth	Decrease	Ruiz et al. (2001)
	Leaf area / shoot size	Decrease	Delgado et al. (1999), Holmer et al. (2008a), Apostolaki et al. (2009), Rountos et al. (2012)
	Rhizome growth	Decrease	Delgado et al. (1999), Marbà et al. (2006)
Associated community	Epiphyte load	Increase	Delgado et al. (1997), Delgado et al. (1999), Pergent et al. (1999), Cancemi et al. (2003)
	Grazing pressure	Increase	Delgado et al. (1997), Delgado et al. (1999), Ruiz et al. (2001), Holmer et al. (2008a), Ruiz Fernandez et al. (2009)

Mediterranean finfish farms are quite different to those in Atlantic Canada because Mediterranean farms are generally situated in low nutrient ('oligotrophic') waters, in very shallow depths (usually 5 – 10 m), close to shore (within 200 – 500 m) in highly sheltered areas. In addition, Neptune grass (specifically *P. oceanica*) has a depth limit of around 40 m (Mayot et al. 2006, Ivana et al. 2020), which is much deeper than the 12 m maximum depth reported for eelgrass (Moore and Short 2006). Consequently, there may be less potential for finfish aquaculture in Atlantic Canada to overlap with / impact eelgrass as ambient water nutrient levels are higher (Powley and Krom 2017) and because sites tend to be located in deeper waters located further from the shore (see Section 6.2.1). It is also highly likely that finfish aquaculture sites in Atlantic Canada are more exposed to waves, wind, tides, and currents which should encourage greater dispersal of dissolved and particulate wastes (reviewed in Howarth et al. 2019).

7.2.2. Evidence from Nova Scotia

To date, only one field study has investigated finfish aquaculture and seagrass interactions outside of the Mediterranean. This was conducted by Cullain et al. (2018) at a finfish farm in Port Mouton Bay, located on the south shore of Nova Scotia, at a depth of 12 m. Eelgrass patches in depths of 1.7 – 2.9 m were surveyed at 300 m, 700 m, and 3 km from the fish farm, and compared to eelgrass beds in the neighbouring bay of Port Joli, and seven other reference areas located on the south and eastern shore of Nova Scotia. Results indicated that eelgrass cover was statistically lower in Port Mouton Bay than the reference areas, and that eelgrass cover exhibited a general declining trend with increasing proximity to the fish farm. Shoot density, and above and below ground biomass also exhibited similar trends but were not statistically significant. Likewise, there was no difference in canopy height or tissue nitrogen content between eelgrass patches near the farm compared to reference areas. All other variables exhibited inconsistent trends. For example, epiphyte cover was substantially higher in eelgrass patches located 700 m away from the farm but was almost non-existent 300 m and 3 km away. A modelling study also suggested a link may exist between anecdotal reports of eelgrass deterioration within the bay and nitrogen effluents emanating from the finfish farm (Cullain et al. 2018). However, a subsequent modelling study concluded that dissolved nitrogen concentrations within Port Mouton Bay during the operation of the fish farm were well below the expected toxicity threshold for eelgrass (Filgueira et al. 2021).

Overall, finfish aquaculture and seagrass interactions are less clear in Port Mouton Bay than studies in the Mediterranean (see Section 7.2.1). Nevertheless, a single field study is insufficient to reach definitive conclusions on finfish aquaculture / seagrass interactions in temperate ecosystems. Thus, further investigation is warranted.

7.3. The effects of shellfish aquaculture on water and sediment biochemistry

Mussels, oysters, scallops, clams and other 'bivalves' feed by pumping in water and filtering out food particles comprising of bacteria, phyto- and zooplankton, detritus, and other organic matter (Newell 2004). After ingestion, particles are sorted, digested, and excreted in the faeces, or ejected as undigested 'pseudofaeces'. Both sink towards the seafloor following release and are collectively referred to as 'biodeposits' (Shumway et al. 1985, Beninger et al. 1999). As biodeposits transfer nutrients from the water column to the seabed, they can increase the nutrient and organic content of sediments underlying shellfish farms (Crawford et al. 2003, Dumbauld et al. 2009). This can lead to enhanced bacterial activity, and where oxygen depletion occurs, an increase in sulphides (Nizzoli et al. 2006, Hargrave et al. 2008, Vinther and Holmer 2008, Richard et al. 2013). Conversely, by removing organic particles from the water column, high bivalve densities can reduce suspended nutrient levels and turbidity, increasing the amount of light reaching the seafloor (Newell and Koch 2004, Ferreira and Bricker 2019, Petersen et al. 2019). Bivalves also excrete nitrogenous wastes (mostly NH_4^+) directly into the water column which can influence coastal nitrogen cycling (Pietros and Rice 2003, Cranford et al. 2007, Ferreira and Bricker 2019).

7.4. Shellfish aquaculture and seagrass interactions

7.4.1. Positive effects

It has been suggested that bivalve aquaculture may have some positive effects on seagrass. This is because bivalves can reduce turbidity and suspended nutrient loads, which can provide more light to seagrasses (Newell and Koch 2004, Ferreira and Bricker 2019, Petersen et al. 2019). Also, their biodeposits can increase sediment nitrogen and phosphorus concentrations, which can provide more nutrients for seagrass growth (Peterson and Heck 2001, Newell and Koch 2004, Dumbauld et al. 2009, Skinner et al. 2014). For example, a study in Baja California, Mexico, found evidence that suspended oyster farms elevated water and sediment NH_4^+ concentrations, which correlated with greater eelgrass shoot size, leaf growth, and photosynthesis (Sandoval-Gil et al. 2016). Similarly, the establishment of a suspended oyster farm in New Zealand correlated with an increase in seagrass cover beneath and adjacent to the farm (Bulmer et al. 2012). Evidence from Japan suggests that oyster farms can also reduce eelgrass epiphyte loads by feeding on suspended benthic diatoms that would otherwise settle upon the eelgrass (Smith et al. 2018). Likewise, field experiments in Florida, USA, found high mussel densities reduced seagrass epiphyte loads and increased sediment nutrient concentrations, resulting in an increase in eelgrass leaf size and growth rates (Peterson and Heck 2001).

7.4.2. Neutral and negative effects

Despite the potential for shellfish aquaculture to benefit seagrass (see Section 7.4.1), most studies suggest shellfish aquaculture has a neutral or negative effect on seagrass. Skinner et al. (2013) surveyed 15 suspended oyster bag farms in Eastern New Brunswick and observed eelgrass meadows with 5 % lower above and below ground biomass within lease sites compared to reference sites 300 m away, and that these differences reached as high as 79 % in some areas. These negative effects were largely limited to a 25 m radius from lease boundaries which quickly diminished with increasing distance form the farms. They also observed that eelgrass growing within lease site boundaries displayed a 38 % reduction in photosynthetic efficiency and capacity, suggesting shading from aquaculture infrastructure was the main factor responsible for these negative trends. Subsequent field experiments supported this notion as shading from oyster cultures reduced eelgrass shoot density, above and below ground biomass, canopy height, leaf size and photosynthetic capacity (Skinner et al. 2014). These negative responses were detected within 67 days after exposure to 26 % subsurface irradiance and exhibited no substantial recovery 253 days after shading treatments were removed.

On a larger scale, a recent meta-analysis examined 125 studies on the effects of shellfish aquaculture on eelgrass (Ferriss et al. 2019). Generally, all methods had negative effects on eelgrass density and biomass, however, the extent of these impacts were highly variable and depended on the production and harvest methods being used. For instance, longline methods negatively impacted eelgrass density, whereas suspended bag aquaculture had a neutral effect. This could be because suspended bag methods allow more light to penetrate to the seafloor. Oyster culturists in new Brunswick, for example, intentionally leave some slack in their lines to allow the bags to move with the tides, reducing physical strain on the gear (Transport Canada 2007, Skinner et al. 2013). This movement would also prevent any areas of the seabed from becoming permanently shaded, potentially permitting the growth of eelgrass. In support of this, dense beds of seagrass have been observed to grow under suspended oyster bags / baskets in Australia (Crawford et al. 2003) and studies have shown that suspended bag aquaculture can result in 68 % less shading than other 'off-bottom' production methods (Madigan et al. 2000).

The meta-analysis by Ferriss et al. (2019) also showed that, for production methods which required workers to harvest shellfish from the sediment (e.g. clam aquaculture and some oyster production methods), mechanical harvesting methods (e.g. dredging, dragging and sediment liquefaction) had the largest initial impact on eelgrass meadows and required the longest time for recovery. Conversely, manual harvest methods (e.g. by hand, or hand tools like rakes and hoes) had less impact on eelgrass, presumably because they can be more spatially targeted, resulting in less disruption to eelgrass roots and rhizomes, and faster recovery times (Cabaço et al. 2005, Wootton and Keough 2016).

8. Aquaculture and eelgrass management in Atlantic Canada

8.1. Regulation in Nova Scotia

Aquaculture is jointly managed by federal and provincial governments, the nature of which varies between provinces (Fisheries and Oceans Canada 2020a). At the federal level, DFO is the primary department responsible for regulating aquaculture through the *Aquaculture Activities Regulations* (SOR/2015-177) which were created under the *Fisheries Act* (R.S.C. 1985). On a provincial level, the *Fisheries and Coastal Resources Act* (S.N.S. 1996) is the primary law governing aquaculture in Nova Scotia, along with the *Aquaculture Licensing and Leasing Regulations* (N.S. Reg. 347/2015) and *Aquaculture Management Regulations* (N.S. Reg. 348/2015). NSDFA are the lead regulators of aquaculture in Nova Scotia and Nova Scotia Environment are responsible for the compliance and enforcement of the *Fisheries and Coastal Resources Act* (Nova Scotia Environment 2020). NSDFA are responsible for issuing aquaculture licenses and leases, performing site visits and other administrative checks, and specifying management measures regarding fish health. Both NSDFA and DFO share the responsibility for environmental management and monitoring (Nova Scotia Department of Fisheries and Aquaculture 2019a).

All aquaculture sites in Canada require a valid lease and licence. The lease entitles the owner or operator to install and use aquaculture gear in a specified area. Whereas the licence allows the owner or operator to stock the facility, subject to conditions specified within the licence. In Nova Scotia, aquaculture license and lease applications undergo either an 'administrative' or 'adjudicative' decision process (reviewed in Nova Scotia Department of Fisheries and Aquaculture 2020d). The administrative decision process is overseen by NSDFA, and covers applications for land-based operations, as well as marine operations regarding experimental licenses and leases, renewals of existing licenses and leases, and amendments to existing licenses and leases that do not result in the expansion of the site or the addition of finfish to a site that is currently not approved for finfish. In contrast, the adjudicative decision process is overseen by the Nova Scotia Aquaculture Review Board (https://arb.novascotia.ca/) and covers applications for new marine licenses and leases, as well as amendments to existing licenses and leases that expand site boundaries or add finfish to the species being cultured.

All aquaculture applications are reviewed by NSDFA and network partners. Network partners consist of a range of provincial and federal departments that have regulatory jurisdiction over certain aspects of aquaculture. This may involve several groups within DFO including the Fish and Fish Habitat Protection Program, Aquaculture Management, the Marine Planning and Conservation program and DFO Science. DFO's fisheries departments may also be involved as mussel and oyster seed collection outside aquaculture leases is considered a fisheries-related activity. Other federal departments like CWS, Transport Canada (TC) and the Canadian Food Inspection Agency (CFIA) may also be involved in the process (Jeffrey Barrell, DFO, pers. comm. 21st July 2020). Together, NSDFA and network partners evaluate the potential environmental impacts of aquaculture proposals based on published scientific research, and physical and ecological data collected during baseline sampling (Nova Scotia Department of Fisheries and Aquaculture 2019b).

Proposed / existing aquaculture operations must undergo baseline / repeated (finfish leases only) environmental monitoring as specified within the federal *Aquaculture Activities Regulations* (Department of Fisheries and Oceans 2018) and provincial Environmental Monitoring Program (Nova Scotia Department of Fisheries and Aquaculture 2020e). These aim to minimize impacts on fish and fish habitat and to allow regulators to impose mitigative measures if an aquaculture operation was deemed to be significantly impacting the marine environment (Nova Scotia Department of Fisheries and Aquaculture 2020b). Exact sampling strategies are determined by a number of factors including sediment composition (i.e. soft or hard bottom), the scale of the operation, and the species being cultured (i.e. shellfish or finfish). For baseline sampling, a number of video transects across the seabed have to be conducted within the proposed lease site, and videos may be required at each corner of the site as well. The raw video footage (and any data reports if required) are then submitted to NSDFA and DFO for review. As the protection of ecosystems and fish habitat falls under the mandate of DFO (Fisheries and Oceans Canada 2021), DFO are responsible for determining if the aquaculture operation poses a risk to eelgrass and other fish habitat, and whether additional mitigation or avoidance measures are needed (Edward Parker, DFO, pers. comm. 4th August 2020). For instance, DFO may: require the operator to place anchors away from sensitive habitats; specify a minimum distance between rows of gear; and / or impose a cap to the maximum area of the lease site they can occupy with gear (Danielle St. Louis, NSDFA, pers. Comm, March 2021).

These evaluations and recommendations are then submitted to the Nova Scotia Aquaculture Review Board (if undergoing an adjudicative application process) who hold an independent tribunal, in which applicants and stakeholders (e.g. members of the public and local industries) present their evidence in support of, or opposition, to the application. The Board then decides whether the application is approved based on appropriate support data presented during the tribunal, and from the recommendations received from NSDFA, DFO and other network partners.

8.2. Aquaculture as a prescribed activity

As discussed earlier (see Section 4.5), section 35 of the *Fisheries Act* legally prohibits a HADD to eelgrass and other fish habitats. However, exceptions are granted to works, undertakings or activities that will result in a HADD if they are 'prescribed' or belong to a 'prescribed class', or if permission is granted by a federal authority (Fisheries and Oceans Canada 2020f). As the installation, operation, maintenance, and removal of aquaculture facilities are prescribed in the *Aquaculture Activities Regulations*, they are excepted providing the conditions in the regulations are met. One of the main conditions is that "reasonable measures" must be taken to avoid and minimize impact to fish habitat (Edward Parker, DFO, pers. comm. 4th August 2020).

8.3. The Bay Management Framework (BMF) in New Brunswick

It is difficult to empirically test whether aquaculture operations directly impact eelgrass beds. Not only are eelgrass beds highly dynamic (see Section 2), provincial and federal regulators typically have access to very little data on eelgrass (such as their location, density, and health) and do not have the resources to perform detailed surveys on a large scale. To overcome these issues, federal and provincial agencies in Eastern New Brunswick have implemented a comprehensive Bay Management Framework (BMF) for suspended oyster aquaculture. The BMF established a broad range of site selection criteria and operating guidelines for suspended oyster aquaculture, which help ensure oyster farms have minimal impact on eelgrass and other species and habitats, while creating no additional data collection requirements on growers and regulators. Provided growers follow these guidelines, regulators assume suspension oyster aquaculture has a negligible impact on eelgrass in Eastern New Brunswick. As the constraints imposed by the BMF are considered more than sufficient to prevent aquaculture from impacting eelgrass, aquaculture does not trigger a HADD, and no environmental assessments are required.

A key reason for the creation of the BMF was to reduce administration loads on growers, as well as federal and provincial agencies. Prior to 2012, proposals for new aquaculture operations and amendments had to undergo an Environmental Impact Assessment (EIA) under the *Canadian Environmental Assessment Act* (S.C. 1992). As the suspended oyster aquaculture industry was undergoing rapid expansion during this time, DFO, TC, and other federal agencies were receiving hundreds of EIAs each year for small (~ 5 ha) suspended oyster farms. Not only did this impose large time and financial costs on growers and federal agencies, the proposed oyster farms were all similar in design, meaning similar environmental effects were expected. Consequently, the EIAs were largely identical and comments from federal agency reviewers were repetitive. Furthermore, the site-by-site application process did not consider cumulative impacts and potential conflicts between multiple coastal users. To address these issues, TC and DFO compiled a 'replacement class screening report' that streamlined the EIA process for suspended oyster farms in Eastern New Brunswick (Transport Canada 2007). The *Canadian Environmental Assessment Act* (S.C. 2012) was later revised in 2012 and no longer required aquaculture projects to undergo an EIA. Therefore, the replacement class screening report was also revised based on lessons learned during the intervening years, and superseded with a Comprehensive Environmental Effects Determination (CEED) report (Transport Canada 2013).

Both reports established a BMF for suspended oyster aquaculture farms on leases managed by the New Brunswick Department of Agriculture, Aquaculture and Fisheries (DAAF). A key component of the BMF was the identification and mapping of all areas in Eastern New Brunswick were deemed suitable for suspended oyster aquaculture. Then, based on existing data and knowledge, a series of buffer zones were created including: a 100 m buffer from the low water mark; 300 m buffers around any conservation areas, species of special concern, migratory birds and fish and their associated habitats; and navigation corridors every 400 m to ensure water users could navigate between leases and still gain access to the shore. Zones were then defined indicating where aquaculture leases already existed, and where new shellfish leases would be best located to protect the environment and avoid conflict with other coastal users. These maps were developed in partnership with a wide range of federal and provincial agencies, and in consultation with a variety of industry, environmental, community and First Nations groups. Overall, this approach represented an early implementation of the principles of Marine Spatial Planning (MSP), which is currently being pursued by federal and provincial regulators for the management of aquaculture and other ocean-based activities across Canada (Fisheries and Oceans Canada 2018). This style of approach is also used by NSDFA and the Municipality of the District of Argyle in the Lobster Bay ADA (see Section 6.1).

The creation of the replacement class screening and CEED documents also involved an extensive consultation process with growers, regulators, researchers, public and other stakeholders. Through these consultations, and by reviewing existing research, it was concluded that shading was the primary impact of suspended oyster aquaculture on eelgrass. Nutrient enrichment of underlying sediment was considered unlikely as all leases were situated in areas with water current velocities greater than 0.2 m s^{-1}, which exceeds the level required to re-suspend and disperse biodeposits (Widdows et al. 1998, Giles and Pilditch 2004). Therefore, the CEED report established a broad range of operating conditions aimed at preventing oyster aquaculture gear from damaging eelgrass through excessive and persistent shading:

- Operators cannot harvest, or knowingly destroy, marine plants;
- To reduce damage to eelgrass and prevent dragging of gear, operators must size anchors appropriately, or install them permanently;
- Anchors should be installed in winter when the effects of turbidity on eelgrass are minimal;
- Gear must be anchored in a way that allows it to sway and move during each tidal cycle;
- Structures should be designed and installed to maximize light penetration to seabed;
- Moorings and other structures are encouraged to be placed away from eelgrass;
- Gear cannot cover more than 50 % of a lease site;
- Rows of gear must be spaced at least 3 m apart; and
- Dead oysters and their shells must be disposed of on land.

DAAF have also capped the total coverage of suspended oyster aquaculture in a single bay to 10 % to help protect eelgrass and other habitats, and to help reduce conflict with other users. Preliminary calculations suggest that the current level of aquaculture in Eastern New Brunswick is likely reducing eelgrass productivity by just 0.1 % but could reach as high as 0.3 % if the 10 % bay limit was reached. Both are considered sustainable and in-line with levels caused by natural variability (Joseph LaBelle, DAAF, pers.com, August 28th 2020).

Overall, the BMF and CEED guidelines are widely accepted by the industry and public. In fact, the oyster aquaculture industry grew by 20 % (by number of bags in the water) between 2018 – 2020, yet no disputes have been raised from the fishing industry, public and / or other stakeholders. The BMF also employs an adaptive management approach where data from any field surveys, and new scientific research, are reviewed on an annual basis to determine if any changes need to be made to the BMF. If the 10 % bay limit is ever reached, provincial and federal regulators intend to launch a new, full investigation into the measures required to ensure these sustainable aquaculture practices continue.

In summary, the BMF is a spatial management system for suspended oyster aquaculture that minimizes user conflicts, environmental impacts, and cumulative effects, without requiring any additional data collection.

9. Management recommendations

The BMF and CEED guidelines adopted in Eastern New Brunswick (see Section 8.3) have proven successful at supporting the growth of the aquaculture industry while ensuring it has minimal impacts on eelgrass. Similar to New Brunswick, there is limited data on the exact distribution and status of eelgrass beds in Nova Scotia (see Section 1.3). A similar management system could therefore offer a potential solution to aquaculture and eelgrass management in Nova Scotia. However, the CEED guidelines only address the potential impacts of suspended oyster aquaculture. Thus, this section proposes several additional measures for other forms of aquaculture in Nova Scotia, and a consultation process to help ensure the CEED guidelines are suitable for a Nova Scotian context.

9.1. Oyster, mussel, and scallop aquaculture

Shading is the primary impact of shellfish aquaculture on seagrass (see Section 7.4.2). Consequently, the CEED guidelines establish a broad range of operational guidelines that help ensure suspended oyster aquaculture does not cause excessive and permanent shading to eelgrass (see Section 8.3). Discussions could be held with oyster growers to determine if any of these CEED guidelines could be implemented in Nova Scotia. Discussions with DFO are warranted given their current recommendation for suspended shellfish aquaculture in Nova Scotia is for growers to keep their lines taut in order to reduce the risk of wildlife entanglement (Jason Naug, DFO, pers. comm, February 2021). This conflicts with the CEED guidelines in New Brunswick which require growers to maintain some slack in their lines, allowing suspended gear to move with tides and preventing any underlying eelgrass from becoming permanently shaded.

Oyster growers could also be encouraged to use suspended bag / cage methods where possible, as these generally have less impact on eelgrass than on-bottom methods (see Section 7.4). An added advantage of suspended methods is that oysters tend to be less susceptible to MSX disease (or 'Multinucleate Sphere X) compared to oysters grown using on-bottom methods (Rod Beresford, Cape Breton University, pers. comm, 19th November 2020). Nevertheless, suspended oyster gear can still come into direct contact with eelgrass and cause a physical disturbance during the winter months if / when growers sink their gear to the seabed (see Section 6.2.3). Discussions could therefore be held with industry to determine whether, in cases where oyster farms have potential to overlap with eelgrass (e.g. in depths < 12 m), growers could sink their gear in deeper areas of their lease to help avoid disturbance to eelgrass. Finally, discussions could be held with growers to determine whether any of the CEED guidelines could be applied to the suspended longline mussel and scallop industry.

Overall, any new siting and operational guidelines may only need to be imposed on shellfish leases in depths of less than 12 m, as eelgrass is highly unlikely to occur beyond this depth (see Section 1.4).

9.2. Intertidal clam aquaculture

Physical disturbance is the primary impact of intertidal shellfish aquaculture on eelgrass (see Section 7.4.2). As the intertidal clam aquaculture industry in Nova Scotia harvests and seeds clams by hand, they should be able to avoid disturbing eelgrass as the disturbance caused by these methods is highly localized. Management measures could restrict the industry from using mechanical harvesting methods given their potential to damage eelgrass and their slow recovery from such disturbances (see Section 7.4.2). As clam aquaculture is intertidal, it should be comparatively easy to determine if proposed clam aquaculture operations coincide with eelgrass, and for growers to find alternative locations if needed. A buffer approach could enable an area of protection around existing eelgrass beds, allowing eelgrass patches the opportunity to undergo seasonal and annual fluctuations in size without coming into direct contact with an operation.

9.3. Open net-pen finfish aquaculture

Nutrient enrichment and the deposition of particulate wastes are the primary impact of finfish aquaculture on eelgrass, and are usually confined to the area directly under fish pens for a radius of approximately 300 m depending on depth and current speeds (see Section 7.2). As eelgrass typically has a maximum depth of 12 m (see Section 1.4), situating finfish farms in depths greater than 12 m deep could help avoid overlap with eelgrass habitat. Locating finfish farms in areas with moderate to high current speeds would also encourage the dispersal of dissolved and particulate wastes (see Section 7.2.1), reducing their potential to elevate nutrient levels in sediments and the water column.

9.4. Consultations and adaptive management

Similar to the creation of the CEED guidelines in Eastern New Brunswick, a thorough consultation process could help ensure proposed guidelines are suitable for Nova Scotia, and agreeable to regulators, industry, and other stakeholders. A regular review process based on stakeholder and regulator feedback, and current scientific research, would help ensure ongoing appropriateness of the guidelines.

9.5. Potential differences between New Brunswick and Nova Scotia

Environmental conditions in the Gulf of St. Lawrence differ from those in Nova Scotia. For example, ice cover and scouring are greater in the Gulf, whereas wave energy is higher in Nova Scotia. There is potential for these differences to cause regional variation in how eelgrass responds to aquaculture (Melisa Wong, DFO, pers. comm, February 2021). Also, the Gulf of St. Lawrence has higher and more continuous eelgrass cover than Nova Scotia, which may make any impacts to eelgrass more ecologically significant in Nova Scotia (Jeffrey Barrell, DFO, pers. comm, March 2021). Nevertheless, the potential impacts of aquaculture on eelgrass are the same - shading by shellfish aquaculture and nutrient enrichment by finfish aquaculture. Consequently, the proposed measures described above are still applicable in protecting eelgrass from aquaculture impacts. Nevertheless, a thorough consultation process with scientists and experts would further ensure the developed guidelines are appropriate for a Nova Scotia context.

9.6. The Aquaculture Review Board

In theory, adopting the proposed guidelines described above, combined with additional research and consultations, could help ensure aquaculture in Nova Scotia has minimal impacts on eelgrass. This could enable the Nova Scotia Aquaculture Review Board (see Section 8.1) to presume that applications for new aquaculture sites and amendments will not harm eelgrass providing the industry follow these developing operational and siting guidelines.

10. Summary

Eelgrass is an ‘Ecologically Significant Species’ and protected under Canadian federal legislation. However, many eelgrass beds in Atlantic Canada have declined in response to a multitude of interacting stressors. As the aquaculture industry continues to grow, there is increasing potential for aquaculture to have a negative impact on eelgrass. Provincial and federal regulators typically have limited access to eelgrass data and do not have the resources to conduct detailed large-scale surveys. This can limit their capacity for evidence-based management of aquaculture and eelgrass interactions. These issues have largely been addressed in New Brunswick due to the adoption of the BMF and CEED guidelines. This management system established a broad range of operating and siting guidelines on the suspended oyster aquaculture industry that have proved effective in minimizing impacts on eelgrass, without the need for additional data collection. A similar management system could therefore offer a potential solution to aquaculture and eelgrass management in Nova Scotia. However, the CEED guidelines only address the potential impacts of suspended oyster aquaculture. Thus, this report proposes several additional measures for other gear types, as well as a consultation process, which could help reduce any potential impacts aquaculture may have on eelgrass in Nova Scotia.

Acknowledgements

Many people assisted in the creation of this report. Thank you to Aaron Ramsay, Alison Schmidt, Brett Painter, Catalina Gomez, Chris Beck, Danielle St. Louis, David Fenwick, Doug Bertram, Duncan Bates, Edward Parker, Gary Pardy, Grace Murphy, Heike Lotze, Jason Naug, Jeffrey Barrell, John Davidson, Kevin Schyf, Lindsay Brager, Matthew King, Melisa Wong, Nathaniel Feindel, Ryan Horricks, Scott Coffen-Smout, Sebastian Haas, Susannah Anderson, and Tim Glasby. We would also like to thank several reviewers for their comments on earlier drafts of the manuscript.

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Vol. 12: 541–557, 2020
https://doi.org/10.3354/aei00381

AQUACULTURE ENVIRONMENT INTERACTIONS
Aquacult Environ Interact

Published December 3

On the edge: assessing fish habitat use across the boundary between Pacific oyster aquaculture and eelgrass in Willapa Bay, Washington, USA

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ABSTRACT: Estuaries are subject to diverse anthropogenic stressors, such as shellfish aquaculture, which involve extensive use of estuarine tidelands. Pacific oyster *Crassostrea gigas* aquaculture is a century-old practice in US West Coast estuaries that contributes significantly to the regional culture and economy. Native eelgrass *Zostera marina* also commonly occurs in intertidal areas where oyster aquaculture is practiced. Eelgrass is federally protected in the USA as 'essential fish habitat', restricting aquaculture activities within or near eelgrass. To contribute scientific information useful for management decisions, we sought to compare fish habitat use of oyster aquaculture and eelgrass, as well as the edges between these 2 habitats, in Willapa Bay, Washington, USA. Furthermore, given a recent shift towards off-bottom culture methods, in part to protect seagrasses, long-line and on-bottom oyster aquaculture habitats were compared. A combination of direct (underwater video, minnow traps) and indirect (predation tethering units, eelgrass surveys) methods were employed to characterize differences in fish habitat use. Eelgrass density declined within both aquaculture habitats but less so within long-line aquaculture. Most fish species in our study used long-line oyster aquaculture and eelgrass habitats similarly with minimal edge effects, and on-bottom aquaculture was used less than either of the other 2 habitat types. These results are consistent with previously observed positive relationships between fish abundance and vertical habitat structure, but also reveal species-specific behavior; larger mesopredators like Pacific staghorn sculpins were sighted more often in aquaculture than in interior eelgrass habitats.

KEY WORDS: Oyster aquaculture · Eelgrass · Habitat · Edge effects

1. INTRODUCTION

As intersections of terrestrial, freshwater, and marine systems, estuaries provide a wide array of ecosystem services and have helped to support flourishing human populations for centuries (Costanza et al. 1997, Lotze et al. 2006). Along the West Coast of the USA, estuarine tidelands have been used extensively for shellfish production, beginning with harvest of native oysters *Ostrea lurida* by native Americans for millennia and by European immigrants since the mid 1800s, shifting towards the current culturing of Pacific oysters *Crassostrea gigas* in the 1920s (Baker 1995, Lindsay & Simons 1997, Robinson 1997, Shaw 1997). Shellfish aquaculture is an economically important practice that supports a diverse industry and provides seafood for people across the country. In Washington State alone, commercial aquaculture of Pacific oysters brought in $32.4 million in 2016 (NMFS 2016). Currently, aquaculture within the USA

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amounts to just 5% of the seafood consumed domestically (NOAA 2011), highlighting the potential for expansion of shellfish aquaculture to help meet this domestic trade gap and rising demand. However, growth of shellfish production is currently partly restricted by regulations put in place to limit potential impacts of aquaculture on other managed estuarine resources and protect other human interests within these estuaries.

Seagrasses provide a variety of ecosystem services and have been globally recognized as important foundation species and ecosystem engineers (Jones et al. 1994, Costanza et al. 1997). These services include coastal protection, global carbon sequestration, and improved water quality (Orth et al. 2006). Most relevant to the current study is their role in providing nursery habitat for juvenile fish and invertebrates (Beck et al. 2001, Heck et al. 2003, Orth et al. 2006, Nagelkerken et al. 2015, Sheaves et al. 2015). The native eelgrass *Zostera marina* of the US West Coast provides habitat for early life stages of commercial species like salmonids, Dungeness crab, rockfish, and English sole (Rooper et al. 2003, Holsman et al. 2006, Dumbauld et al. 2015, Olson et al. 2019). This clear but indirect connection between eelgrass and the economic success of fisheries is the reason for its protection in the USA as 'essential fish habitat' under the Magnuson-Stevens Fishery Conservation and Management Act (16 USC §§1801–1891d). This designation prohibits the damage or destruction of eelgrass and forces consideration of trade-offs with the permitting of other interests, such as shellfish aquaculture. Under current regulations implemented by the Pacific Fishery Management Council, existing aquaculture is generally permitted to continue as practiced, but new aquaculture is prohibited within 25–30 feet (7.6–9.1 m) of existing eelgrass beds (Pacific Fishery Management Council 2014).

Eelgrass and oyster aquaculture occur at similar tidal elevations and often overlap within relatively large areas in US West Coast estuaries (Dumbauld & McCoy 2015). While these 2 habitats are not mutually exclusive, aquaculture practices can sometimes limit the extent of eelgrass (Wisehart et al. 2007, Tallis et al. 2009, Wagner et al. 2012, Skinner et al. 2013). Oyster aquaculture can result in both long-term disturbances, such as the addition of oysters and associated gear, and short-term disturbances, like harvest or maintenance events, that can impact the quantity and quality of eelgrass habitat within estuaries (Simenstad & Fresh 1995, Dumbauld et al. 2009). Characterization of habitat use of both oyster aquaculture and eelgrass could help to further inform future management decisions and marine spatial planning concerning potential conflicts between these 2 uses.

Estuarine habitats with more habitat structure have generally been shown to support higher faunal abundances and diversity (Orth et al. 1984, Jenkins et al. 1997, Heck et al. 2003). Eelgrass provides such a natural biogenic structure and has been widely documented to harbor a more diverse assemblage of organisms compared to unvegetated areas (Ferrell & Bell 1991, Pinnix et al. 2005, Hosack et al. 2006, Ferraro & Cole 2007, Gross et al. 2017). Oysters and oyster aquaculture also create structured habitat that generally supports a higher diversity and abundance of organisms than adjacent open mudflat (Castel et al. 1989, Pinnix et al. 2005, Hosack et al. 2006, Ferraro & Cole 2007). While higher densities of benthic invertebrates have been found in eelgrass than in oyster aquaculture beds, the abundance of fish and other mobile nekton has largely been shown to be similar or even greater within aquaculture beds (DeAlteris et al. 2004, Pinnix et al. 2005, Hosack et al. 2006).

Importantly, the arrangement of different habitats at landscape spatial scales (100s–1000s of m^2) also affects the distribution of organisms (Forman & Godron 1986, Turner 1989, Wiens & Milne 1989), but this approach has rarely been taken when examining seagrass–aquaculture interactions. Furthermore, edges or boundaries between habitats, which may result in abrupt changes in resource availability, refugia, and predation pressure, can strongly influence the abundance and diversity of organisms (Gates & Mosher 1981, Sisk & Haddad 2002, Ewers et al. 2007). In marine ecosystems, seagrasses have been a focal system for research on edge effects because of their natural propensity to form discrete patches (Boström et al. 2006, 2011). Patterns of faunal abundance are complicated at these habitat edges, where greater densities of organisms can occur compared to the core habitat (Bologna & Heck 2002, Tanner 2005, Smith et al. 2008), yet the opposite relationship has also been reported (Bell et al. 2001, Jelbart et al. 2006). Inconsistent seagrass edge effects could be related to several factors including species characteristics (Eggleston et al. 1998, Smith et al. 2010), patch size (Bowden et al. 2001, Smith et al. 2010), habitat complexity (Hovel & Lipcius 2001, Pinna et al. 2013), body size and life stage of organisms (Hovel & Lipcius 2001, Selgrath et al. 2007), and the sharpness of the habitat transition (Matias et al. 2013). These factors in turn influence the strength of edge effects by changing resource availability and predation pressure. Due to the propensity of oyster aquaculture and eelgrass to overlap and regulatory constraints placed on expansion of shellfish operations to avoid this overlap, edge effects between these 2 habitats are a pertinent question for managers of US West Coast estuaries. Furthermore, growth of the oyster aquaculture industry could mean an increase in the number or size of aquaculture/eelgrass boundaries. Thus, information about use of the edge habitat between aquaculture and eelgrass is necessary to inform management and regulation.

Investigation into the impact of edges between aquaculture and eelgrass beds on faunal abundance is further complicated by the wide range of aquaculture methods used in US West Coast estuaries. These methods differ in many characteristics, including the habitat structure created and harvest method. To date, research has focused on on-bottom (OB) culture methods, because this has historically been the primary technique for growing oysters in most estuaries along the US West Coast (Dumbauld et al. 2009). However, off-bottom culture is becoming increasingly popular due to regulatory constraints and market trends. This method can result in a higher-quality product for the half-shell market (Walton et al. 2012) and has also been shown to reduce some impacts to eelgrass, as disturbance due to mechanical harvesting is reduced (Tallis et al. 2009, Ferriss et al. 2019). The ecological impacts of such practices, where cages, floats, rafts, lines, and supporting structures are also placed in the estuary, are less well-understood. The habitat provided by these new types of aquaculture is distinct from that provided by oysters alone in OB aquaculture. Comparing OB aquaculture habitat with habitat created by off-bottom aquaculture provides additional information about the impact of this industry on the estuarine habitat matrix.

In this study, we explored the similarities and differences between fish habitat use of oyster aquaculture and eelgrass by addressing 2 main questions: (1) Do oyster aquaculture and eelgrass habitats support different abundances of fish, and is there an associated effect at the edge between these habitats? (2) Does the aquaculture method affect the difference seen amongst habitats (if any)? Together, these questions were designed to provide an ecological basis for an integrated framework of management regulations related to the overlap of eelgrass and oyster aquaculture in US West Coast estuaries.

Ecological theory suggests that habitat structure increases faunal abundance and diversity (Orth et al. 1984, Jenkins et al. 1997, Heck et al. 2003). We posited that differences between aquaculture and eelgrass habitats might only be detectable for OB culture since it provides less vertical structure than eelgrass. In contrast, off-bottom aquaculture habitat might support comparable faunal abundances to those found in eelgrass habitat due to the similarity in vertical habitat structure. Because edge effects typically occur in locations with food–risk tradeoffs (Macreadie et al. 2010, 2012, Smith et al. 2011), we anticipated 3 potential trends in abundance at the aquaculture–eelgrass edge: (1) fish abundance could be enhanced if food resources were enhanced, yet risks of being preyed upon were not, (2) fish abundance could be decreased if risks of being preyed upon were higher, and (3) no difference might be observed if both bordering habitats had similar effects. Again, based on the amount of structure in each habitat, we expected edge effects to be most apparent between OB culture and eelgrass. By investigating use of the transition between aquaculture and eelgrass habitats, a more informed and balanced management approach can be reached.

2. MATERIALS AND METHODS

2.1. Study sites

Willapa Bay is a macrotidal estuary located in southwestern Washington State, USA (46.5395° N, 123.9888° W). It is the third largest estuary on the US Pacific Coast, with an area of 358 km^2. The bay is strongly tidally influenced, with about 60% of the total area (215 km^2) considered intertidal (Hedgpeth & Obrebski 1981, Dumbauld & McCoy 2015). Of that 215 km^2, 8% is devoted to oyster aquaculture (17 km^2) and recent surveys have shown that *Zostera marina* occupies approximately 32% of the tide flat (60–80 km^2) (Dumbauld & McCoy 2015). Eelgrass substantially overlaps with oyster aquaculture (13%) and is often found at similar tidal elevations (Ruesink et al. 2006, 2010). OB oyster aquaculture involves spreading oysters set on cultch across the tideflat and harvesting either by dredging or by hand after approximately 3–5 yr of grow-out. The method of off-bottom aquaculture primarily used in Willapa Bay is long-lines (LLs), where oysters are woven into a line that is stretched in rows across the tideflat and suspended on PVC pipe about 0.5 m off the bottom. Our sampling of off-bottom aquaculture habitat focused on this method.

Sampling was undertaken at 3 sites within the bay (Fig. 1): Russell Channel (46.65705° N, 123.94678° W), Tokeland (46.71718° N, 123.94484° W), and Nemah

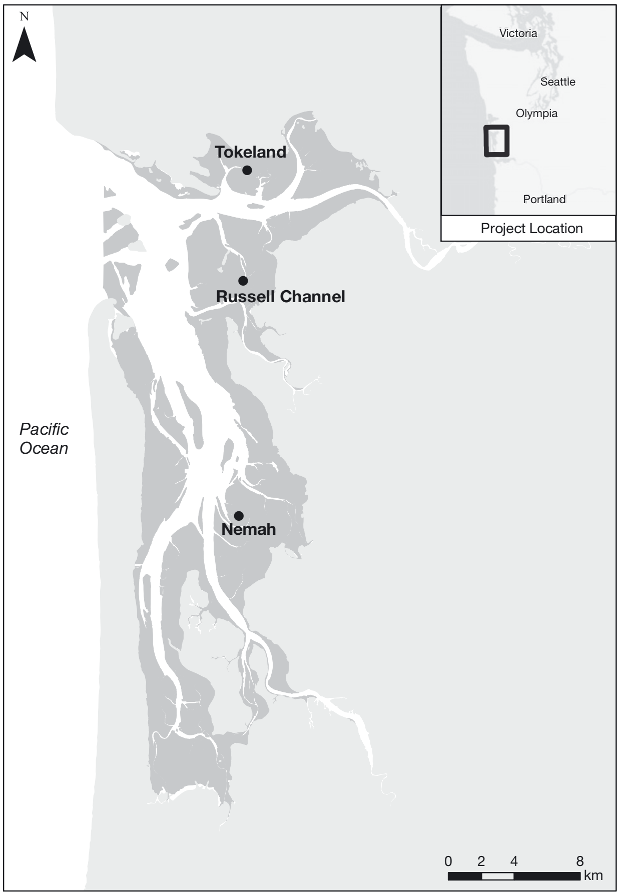

Fig. 1. Sampling sites within Willapa Bay, Washington, USA. Each site included off-bottom and on-bottom aquaculture adjacent to an established eelgrass bed

(46.52833° N, 123.94814° W). These sites were chosen based on 4 factors: (1) presence and appropriate configuration of necessary habitats (OB aquaculture, LL aquaculture, and eelgrass), (2) similar tidal elevation within each site, (3) structural consistency of oyster and eelgrass habitats (e.g. similar density of eelgrass, similar size of oysters), and (4) accessibility at low and high tide. Sampling was undertaken twice at each site: once in July 2017, and a second time approximately 4 wk later in August 2017. Sampling was constrained to the summer months to target the season with peak eelgrass density and also fish abundance and diversity (Orth & Moore 1986, Thom et al. 2003, Hosack et al. 2006, Ruesink et al. 2010, Zhang et al. 2016, Gross et al. 2019).

2.2. Sampling design

Samples were taken along a 60 m transect that was set up perpendicular to the boundary between the aquaculture and eelgrass habitats at each site. This transect was aligned parallel to the nearest channel when possible. At one site (Russel Channel), eelgrass density did not allow for this directionality, resulting in a slightly larger elevation gradient along the transect than at the other 2 sites. An array of sampling methods was used to characterize the differences in species presence and behavior at each of 5, evenly spaced positions (15 m apart) along the transect (Fig. 2). These 5 positions were considered to represent different parts of the habitat matrix: (A) aquaculture interior, (B) aquaculture intermediate, (C) edge, (D) eelgrass intermediate, and (E) eelgrass interior. The edge was defined as the point where aquaculture ceased. This was straightforward for LLs (simply where the culture lines ended). For OB aquaculture, however, the edge was more diffuse and was designated by visually assessing the location where the density of oysters dramatically decreased. The edge was always the middle of the transect, so the interior habitats were each located 30 m into the respective habitat.

2.2.1. Environmental data

Four HOBO® Onset Data Loggers UA-002-64 were used to measure water temperature (°C). One logger was attached to a minnow trap (Fig. 2) deployed in the interior of each aquaculture bed, the interior of the eelgrass bed, and the edge along the LL transect (4 loggers total at each site). Loggers recorded data at 15 min intervals and were used to assess environmental differences in temperature between sites.

2.2.2. Eelgrass sampling

Eelgrass metrics were collected every 3 m along each transect, resulting in a total of 21 data points at each site. At each sampling location, a 0.0625 m^2 quadrat was used to assess percent cover and shoot density of *Z. marina*. Eelgrass morphology was measured on 10 eelgrass shoots from each of the 5 main sampling positions along the transect. These shoots were placed in a cooler and stored at −20°C until processing. The length and width of the longest blade and epiphyte load (dry mass of epiphytes / dry mass shoot) were measured (Hayduk et al. 2019). Length and width were then multiplied together to determine blade surface area. Length was defined as the distance from the last nodule on the rhizome to the end of the blade. Width was measured at the center

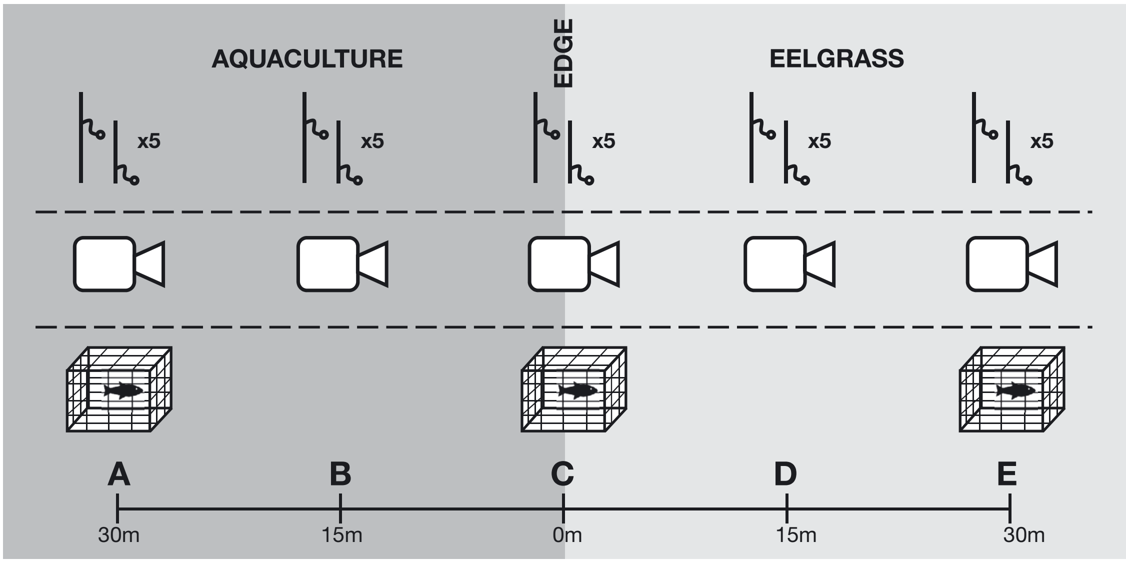

Fig. 2. Sampling design showing predation tethering units (top), cameras (middle), and minnow traps (bottom). Letters refer to main habitat points along the transect (A: aquaculture interior; B: aquaculture intermediate; C: edge; D: eelgrass intermediate; E: eelgrass interior)

of the blade's length. Epiphyte load was determined by scraping the epiphytes off the blade using a microscope slide and then drying the blades and epiphytes separately in an oven at 60°C for 48 h or until a constant weight was reached (Hayduk et al. 2019). To estimate the total epiphyte biomass across the transect for analysis, epiphyte load was multiplied by shoot density. Blade surface area was also multiplied by shoot density to approximate emergent surface area across the transect.

2.2.3. Digital video

Digital video data was gathered using GoPro HERO4® cameras placed at each of the 5 main positions along the transect (Fig. 2). Camera mounts were constructed out of 1″ (2.5.cm) PVC pipe, which included an arm for the camera and a 0.25 m^2 quadrat that lay on the bottom (Fig. S1 in the Supplement at www.int-res.com/articles/suppl/q012p541_supp.pdf). The camera was affixed approximately 30 cm off the bottom and a makeshift Secchi disk was mounted on one corner of the quadrat, approximately 1 m away from the camera. The Secchi disk was used for a quantitative analysis of the visibility in the video using image analysis software. Its placement also acted as a point of reference beyond which organisms were not counted.

The cameras were deployed by snorkeling out to a buoy that had been placed at low tide and were retrieved from a boat approximately 2 h after deployment (when the cameras ran out of battery).

2.2.4. Predation tethering units

Predation intensity was measured with predation tethering units (PTUs) (Duffy et al. 2015, Reynolds et al. 2018). PTUs are used widely in a variety of systems to estimate how much predation is occurring within a given habitat or area. Bamboo stakes with small pieces of dried squid attached as bait were placed at the 5 main positions along the transect, and the presence or absence of the squid was recorded at predetermined time points. Two different PTU treatments ('high' and 'low') were deployed. Dried squid bait (diameter = 0.5 in [1.27 cm]) was superglued to a 10 cm monofilament line and tied at 30 cm above the substrate for the high treatment and 10 cm above the substrate for the low treatment, so that the bait was suspended 20 cm above and just above the substrate, respectively. These 2 treatments were designed to assess different types of predators: those within the water column and those that were searching for prey along the bottom. Five PTUs of each treatment were deployed opposite the traps (Fig. 2) at each position at low tide. The stakes were placed in 2 rows approximately 2 m apart, alternating high and low treatment within each row starting at about 4 m from the transect tape, so as to not influence other sampling techniques. Presence of the squid bait was checked once the water had reached a depth of about 30 cm and then again approximately 24 h later.

2.2.5. Minnow traps

Minnow traps (approximately 60 × 60 × 46 cm, with a ~15 mm opening) were used to sample the fish species and were placed about 5–6 m from the transect tape opposite the PTUs at each of 3 transect positions: A, C, and E (Fig. 2). Traps (un-baited) were deployed at low tide and retrieved approximately 1 h after the local high tide. Captured fish were identified to species (where possible), counted, measured, and then returned to the water.

2.3. Video processing

Video footage was first assessed for visibility using the difference in the pixel values between the black and white quadrants of the Secchi disk. As turbidity increases, the contrast between these 2 sections of the image should decrease to zero. ImageJ (https://imagej.nih.gov/ij/index.html; Schindelin et al. 2015) was used to assess the contrast in a still photo taken every 20 min from each 2 h video (n = 5). Within the software, a horizontal line was drawn from one quadrant of the disk to the other and the grayscale values along this line were exported. The minimum and maximum of the second derivative of the curve were determined and used as the bounding points of the quadrants to obtain average values for the white and black sections of the Secchi disk. These averages were then subtracted to get a contrast for the given image. Grayscale pixel values are assessed on a range from zero (black) to 255 (white), so a maximum contrast would be 255, although this value would not be realistic in natural conditions. In some cases, eelgrass limited the view of the Secchi disk, so as many measurements as possible were made. Only 2 videos did not have any usable images. The average of the calculated contrast values provided a water clarity score for each video. Obstruction by eelgrass or macroalgae was also assessed at each 20 min time point and the video was given an average score based on the percentage of the field-of-view that was blocked. The frame was roughly divided into thirds, and obstruction was assessed in these increments. Correlation between visibility and the number of fish seen was tested using these scores prior to running statistical analyses.

Analysis of the species composition and behavior within each video was completed using BORIS, a free behavioral coding software (Friard & Gamba 2016). Previous experience with video quality suggested that the middle hour of video was appropriate for analysis (Clarke 2017). Thus, observation began at 30 min into the recording and ended at 1.5 h. Within the software, any fish or crab sighting was logged with the species identification and behavior category. Clarke (2017) described 4 behavior categories that were applied to each fish or crab sighted: transit (movement through the frame with no other detectable behaviors), forage (action to ingest or seek out food), school (2 or more fish of the same species moving together, sensu Keenleyside 1955), and refuge (using structure to hide from predators). Because it was difficult to know if an individual reentered the frame once it had left, our response variable is termed as 'sightings', rather than counts. For individuals whose species was unidentifiable, the observation was recorded as such and included in the calculation of total sightings. All videos were watched by the same individual to decrease observer bias. Due to issues with video quality and inconsistencies in the camera gear used for video data collection between the 2 sampling trips, only video from the August 2017 trip was included in this analysis (30 videos, 2 h each).

2.4. Statistical analyses

All data analyses were completed using R v.3.3.1 (R Core Team 2016). Generalized linear mixed models (GLMMs; Bolker et al. 2009) were fit to assess the significance of the position along the transect and aquaculture type for each response variable (see Table 1). Both transect position (i.e. the sampling position along the transect between oyster aquaculture and eelgrass habitats; 5 levels: A, B, C, D, or E) and aquaculture type (2 levels: LL or OB) were treated as categorical fixed effects. When data were available for both time points, sampling date was included as a random effect (2 levels: July or August). Thus, the data structure was a 2-factor design with 3 replicates (site) and a random effect of date (n = 3). These models were fit using the 'lme4' package (Bates et al. 2015). After checking the reasonability of assumptions by examining residuals and leverage of the data, all data (eelgrass survey parameters, epiphyte load, etc.) were fit using a Gaussian distribution, except for PTU, minnow trap, and video sighting data. Count data from the minnow traps and video were fit using a Poisson distribution. Presence/absence data from the PTUs were analyzed using logistic regression with 2 additional fixed effects:

treatment (2 levels: high and low) and check time (2 levels: first and 24 h). GLMM analysis was followed with a Type II Wald χ^2-test (analysis of deviance) using the Kenward-Roger degrees of freedom approximation to assess overall significance of the factors (Schaalje et al. 2002). Because video data were only available for one timepoint, models were run without the random effect of date.

When the interaction term between transect position and aquaculture type was significant in the original model, multiple pairwise comparisons were completed to explore factors driving the significant interaction. Simultaneous *z*-tests were run to examine all pairwise comparisons between the 10 habitats (LL-A, OB-A, LL-B, etc.) using the 'multcomp' package (Hothorn et al. 2008, Wright et al. 2014). Comparisons of interest included those between the edge and other habitats within each aquaculture type, in addition to differences between the aquaculture types at each transect position. Although all pairwise comparisons were made, only significant comparisons of interest are indicated in associated figures to highlight these differences.

3. RESULTS

3.1. Eelgrass survey and environmental data

Both percent cover and shoot density of *Zostera marina* varied with transect position and the interaction term between transect position and aquaculture type (Tables 1 & S1–S4, Fig. S2). Post hoc analyses (Table S2) indicated that eelgrass percent cover in the 2 eelgrass habitats on the LL transect (LL-D and LL-E) was significantly greater than that on the LL edge (LL-C) by a factor of about 2 (i.e. approximately 50 vs. 25%). Eelgrass cover in the OB eelgrass habitats (OB-D and OB-E) was also significantly greater than that at the edge (OB-C) by a factor of 2 (Table S2). Within the OB aquaculture, the intermediate habitat (OB-B) harbored significantly less eelgrass than the OB edge (OB-C; Table S2). Similar patterns were seen in post hoc analyses of the shoot density data (Table S4); shoot densities in the eelgrass habitats along the OB transect (OB-D and OB-E) were significantly greater (averaging about 70 shoots m^{-2}) than those at the OB edge (OB-C; with mean density of 20 shoots m^{-2}), but there was no significant difference between density in these habitats on the LL transect (Table S4). Shoot density at the LL edge (LL-C) differed significantly from density in the aquaculture habitat 30 m from the edge (LL-A), showing an average shoot density about 4 times higher than in the interior of the aquaculture bed (Table S4). The 2 aquaculture types also differed in shoot density at the intermediate eelgrass habitat (D) (Table S4).

Blade surface area varied significantly with transect position, aquaculture type and their interaction (Table 1). Post hoc analyses indicated a significant difference in blade area between the LL edge (LL-C) and LL eelgrass habitat 30 m from the edge (LL-E), as well as between the 2 aquaculture types at the eelgrass interior (OB-E and LL-E; LL-C and LL-E:

Table 1. Analysis of deviance tests following generalized linear mixed models on eelgrass *Zostera marina* presence and structure, minnow trap data, and digital video data metrics. N = 3 for all tests. *p < 0.05

Metric	Fixed effect		
	Transect position	Aquaculture type	Interaction
Eelgrass (*Z. marina*)			
Percent cover	$\chi^2(4) = 240.54$ $p < 0.001$*	$\chi^2(1) = 0.00$ $p = 0.952$	$\chi^2(4) = 13.08$ $p = 0.011$*
Shoot density	$\chi^2(4) = 104.70$ $p < 0.001$*	$\chi^2(1) = 0.26$ $p = 0.610$	$\chi^2(4) = 25.14$ $p < 0.001$*
Blade surface area	$\chi^2(4) = 25.80$ $p < 0.001$*	$\chi^2(1) = 15.11$ $p < 0.001$*	$\chi^2(4) = 12.56$ $p = 0.014$*
Epiphyte load	$\chi^2(4) = 7.69$ $p = 0.104$	$\chi^2(1) = 0.49$ $p = 0.485$	$\chi^2(4) = 6.00$ $p = 0.199$
Emergent surface area	$\chi^2(4) = 35.15$ $p < 0.001$*	$\chi^2(1) = 0.18$ $p = 0.672$	$\chi^2(4) = 5.44$ $p = 0.246$
Total epiphyte biomass	$\chi^2(4) = 18.44$ $p = 0.001$*	$\chi^2(1) = 0.13$ $p = 0.716$	$\chi^2(4) = 6.53$ $p = 0.163$
Traps			
Total catch abundance	χ^2 (2) = 0.45 $p = 0.799$	χ^2 (1) = 7.62 $p = 0.006$*	χ^2 (2) = 11.94 $p = 0.003$*
Species richness	χ^2 (2) = 0.38 $p = 0.827$	χ^2 (1) = 0.12 $p = 0.732$	χ^2 (2) = 1.91 $p = 0.385$
Video			
Total sightings	χ^2 (4) = 24.79 $p < 0.001$*	χ^2 (1) = 57.16 $p < 0.001$*	χ^2 (4) = 36.35 $p < 0.001$*
Species richness	χ^2 (4) = 1.68 $p = 0.794$	χ^2 (1) = 0.15 $p = 0.700$	χ^2 (4) = 1.09 $p = 0.859$
Shiner perch sightings	χ^2 (4) = 38.41 $p < 0.001$*	χ^2 (1) = 62.01 $p < 0.001$*	χ^2 (4) = 39.23 $p < 0.001$*
Pacific staghorn sculpin sightings	χ^2 (4) = 70.59 $p < 0.001$*	χ^2 (1) = 14.28 $p < 0.001$*	χ^2 (4) = 19.13 $p < 0.001$*
Transiting behavior	χ^2 (4) = 25.18 $p < 0.001$*	χ^2 (1) = 55.13 $p < 0.001$*	χ^2 (4) = 55.69 $p < 0.001$*
Foraging behavior	χ^2 (4) = 13.82 $p = 0.008$*	χ^2 (1) = 0.00 $p = 0.948$	χ^2 (4) = 2.71 $p = 0.608$

$z = 4.564$, $p < 0.001$, transect position E: $z = -3.452$, $p = 0.020$). In both cases, blade area at LL-E was significantly greater than that in the habitat to which it was compared. When multiplied by shoot density to estimate emergent surface area provided by eelgrass, transect position was significant (Table 1, Fig. 3). Emergent surface area generally increased from aquaculture habitats into eelgrass beds.

No significant difference in epiphyte load was detected across transect position or by aquaculture type (Table 1). However, as with blade surface area, multiplying epiphyte load by shoot density to estimate the total epiphyte biomass across the transect resulted in only transect position being significant, with higher total epiphyte biomass present in eelgrass habitat than in aquaculture (Table 1).

Average water temperature was approximately 18°C, with little variation in time or space (SE = 0.35°C). The water was about 1°C warmer in August than July (data from only 2 sites in July).

3.2. Digital video

Videos had an average Secchi contrast score of 14.6, with values ranging from 0 to 23.9 (SE = 0.79). While the values themselves do not have any practical meaning, they give a sense of the range of visibility among the videos. The low values (~0–8) represent videos in which the Secchi disk was barely visible at the 1 m distance. Videos with higher than average values have relatively clear visibility to a depth of field of 1 m and even slightly beyond. Videos were about 25% obstructed by eelgrass and algae on average, with values ranging from 0 to 67% (SE = 0.35). Neither visibility metric was highly correlated with total fish sightings in a given video (Secchi contrast: $R = 0.07$, eelgrass obstruction: $R = 0.17$) and were therefore not included in subsequent models.

Ten species of fish and crab were positively identified in 1299 sightings, with an additional 191 sightings in which no identification could be made (Table 2). Analysis of total fish and crab sightings in video data revealed that both transect position and aquaculture type and their interaction were significant (Tables 1 & S5, Fig. 4). Pairwise comparisons of the interaction (Table S6) showed that sightings in the LL aquaculture habitats (LL-A and LL-B) were both statistically greater than those for OB habitats (Fig. 4). For LLs, edge effects were detected, as sightings in habitats 30 m (LL-A) and 15 m (LL-B) into aquaculture and 30 m (LL-E) into eelgrass were all significantly greater than those at the LL edge (Fig. 4, Table S6). No significant difference in sightings of fauna was found between the edge and the other habitats for OB aquaculture. There were also no significant differences in species richness among the transect positions or between aquaculture types (Table 1).

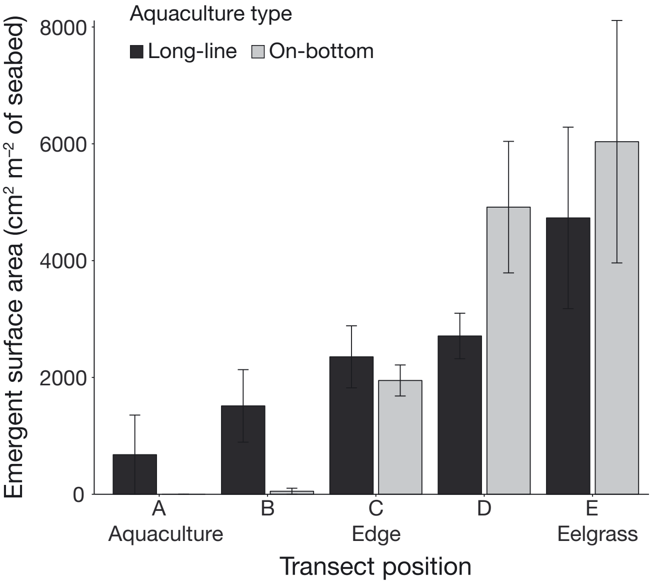

Fig. 3. Emergent surface area of *Zostera marina*, as calculated by blade surface area × shoot density for each aquaculture type and position along the transect (N = 6). Error bars: ±1 SE

Table 2. Species of fish and crabs sighted in underwater video footage and caught in minnow traps. Average size (SE) is also given for those measured in traps. Note: there were an additional 191 sightings that were unidentified. TL: total length; CW: carapace width

Taxon	Common name	Video sightings	Trap catch	Average size
Cymatogaster aggregata	Shiner perch	675	5	43 (1.2) mm TL
Leptocottus armatus	Staghorn sculpin	138	19	131 (2.3) mm TL
Gasterosteus aculeatus	Three-spine stickleback	76	144	
Metacarcinus magister	Dungeness crab	30	50	26 (1.3) mm CW
Rhacochilus vacca	Pile perch	29	0	
Sygnathus leptorhyncus	Bay pipefish	13	7	
Parophrys vetulus	English sole	6	3	94 (3.6) mm TL
Pholis ornata	Saddleback gunnel	5	38	
Ophiodon elongatus	Lingcod	1	0	
Aulorhyncus flavidus	Tubesnout	1	0	
Hemigrapsus oregonensis	Yellow shore crab	0	3	
Pagurus spp.	Hermit crab	0	1	

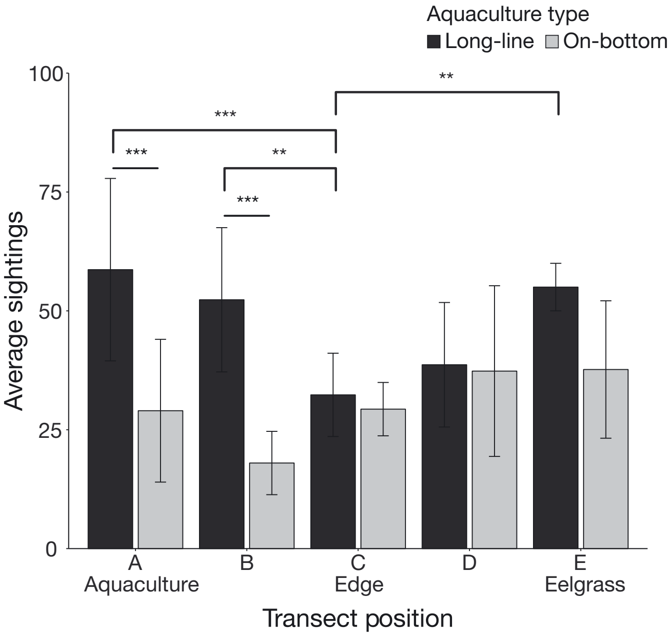

Fig. 4. Average (±1 SE) total sightings of fish and crabs from video data across transects and between aquaculture types (N = 3). Brackets show significant contrasts of interest from simultaneous comparisons of all pairs (significance: $^{**}p < 0.01$; $^{***}p < 0.001$)

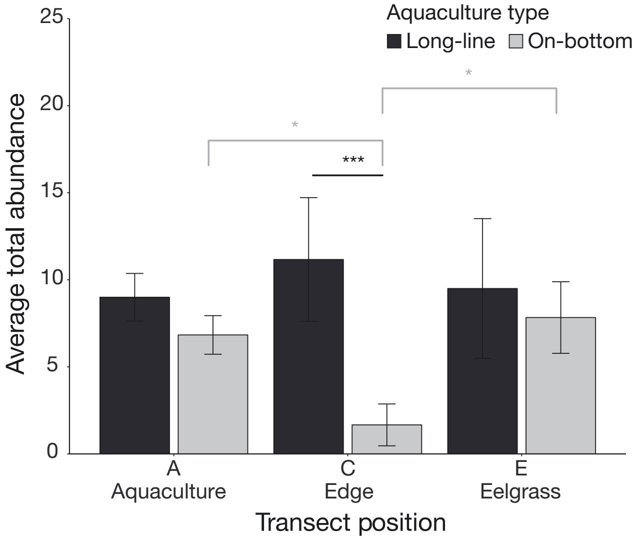

Fig. 5. Average (±1 SE) catch of fish and crabs in minnow traps across the transects and between aquaculture types (N = 6). Brackets show significant contrasts of interest from simultaneous comparisons of all pairs significance: $^{*}p < 0.05$; $^{***}p < 0.001$

Shiner perch *Cymatogaster aggregata* (58% of sightings) and Pacific staghorn sculpin *Leptocottus armatus* (12% of sightings) sightings were each significantly related to both transect position and aquaculture type, along with their interaction (Table 1, Figs. S3 & S4). Dungeness crab *Metacarcinus magister* was the fourth most observed species in video and caught in traps (Table 2). They appeared most often in aquaculture habitats, especially OB culture (Fig. S5), but were not observed consistently enough to allow for informative statistical analyses. Post hoc comparisons indicated that shiner perch sightings in both LL aquaculture habitats (LL-A and LL-B) were significantly greater than in OB habitats (OB-A and OB-B; transect position A: $z = -5.173$, $p < 0.001$, transect position B: $z = -6.037$, $p < 0.001$). In addition, sightings at the OB edge (OB-C) were significantly greater than those at the habitat 15 m into the aquaculture bed (OB-B; OB-B and OB-C: $z = 4.188$, $p < 0.001$). For Pacific staghorn sculpins, sightings in the LL habitat 15 m into the bed (LL-B) were significantly greater than in OB aquaculture (OB-B), but this was the only habitat where there was a significant difference (transect position B: $z = -3.512$, $p = 0.013$). On the LL transect, sightings in both of the aquaculture habitats (LL-A and LL-B) were significantly greater than those at the edge (LL-C; LL-A and LL-C: $z = -3.850$, $p < 0.01$, LL-B and LL-C: $z = -4.241$, $p < 0.001$).

Four behaviors were characterized for fish and crabs in the video: foraging, seeking refuge, schooling, and transiting. By far, the most common behavior observed was transiting (88% of sightings). Both transect position and aquaculture type and their interaction were found to significantly affect the number of transiting sightings (Table 1), but transiting behavior tended to be lowest along the edge. This is consistent with the analysis of the complete data set. Post hoc analyses revealed the same patterns as the data set for all behaviors combined. Analysis of foraging observations (8% of sightings) revealed that transect position was significant (Table 1), and again, foraging behavior trended highest in the structured habitats. Neither of the other behaviors were observed frequently enough to enable separate analyses.

3.3. Minnow traps

Nine species of fish and crab were caught in the minnow traps and mostly confirmed those observed in the video (Table 2). Both aquaculture type and the interaction between the 2 fixed factors were found to significantly affect the total number of fish caught (Tables 1 & S7, Fig. 5).

Pairwise comparisons indicated a significant difference between total catch of fish and crab in aquaculture types at the edge, with the LL edge (LL-C) being greater than the OB edge (Table S8), while there was no discernable difference between catch in eelgrass and aquaculture eelgrass habitats. The high average catch seen at LL-C was mainly driven by a large catch of three-spined sticklebacks *Gasterosteus aculeatus* at one site (Tokeland, 36 in LL-C traps across both sampling points). Evidence of edge effects were most apparent in the OB aquaculture, as catches in both the aquaculture and eelgrass habitats were significantly greater than those at the edge (Table S8). No significant differences in species richness were detected across the transect or between aquaculture types (Table 1).

3.4. PTUs

Both treatment and check time significantly affected the presence of bait (Table 3), with the low treatment being preyed upon more and almost all of the bait absent at the 24 h check. Aquaculture type was also significant, with more predation in LL habitats, while there was no significant interaction with transect position (Fig. 6, Table 3).

4. DISCUSSION

The structural complexity of estuarine habitats is an important factor influencing the diversity and abundance of fauna, including fish and mobile invertebrates. Seagrass beds generally harbor higher diversity and abundance of these organisms versus open unstructured soft bottom habitat (Orth et al. 1984, Hughes et al. 2002, Horinouchi 2007, Ferraro & Cole 2010). Patterns observed are often dependent on species, functional group, or even sampling device (Gross et al. 2017, 2018). Here, we sought to characterize the function of 2 forms of shellfish aquaculture as another structured habitat utilized by nekton and compare that with adjacent vegetated eelgrass *Zostera marina* habitat, with consideration of the edge habitat created between them. We found that fish use of off-bottom LL oyster aquaculture habitat was similar to that of eelgrass. In contrast, average sightings of fish in OB oyster aquaculture were significantly fewer than sightings in both LL aquaculture and eelgrass, which were statistically indistinct (Table 1, Fig. 4). These results may be related to differences in habitat structure, since there is often a positive relationship between habitat structure and fish abundance (Orth et al. 1984, Ferraro & Cole 2010), and both LL oyster aquaculture and eelgrass beds have more habitat structure than OB oyster aquaculture (Fig. 3). Predation intensity measurements and observations of foraging activity in the video footage suggest that LL aquaculture supported predation and foraging rates analogous to those measured in eelgrass. Fish abundance at the edge between the LL aquaculture and eelgrass habitats was generally intermediate but was distinct at the edge between OB culture and eelgrass. This consideration of edges as separate habitat is especially relevant should shellfish aquaculture be expanded in these estuaries. Aquaculture type can affect the distribution of fish across the habitat transition into eelgrass-dominated habitat.

Table 3. Analysis of deviance results following a generalized linear mixed model fit for the predation tethering unit data (N = 3). **Bold** indicates significance ($p < 0.05$)

Fixed effect	χ^2	df	Pr ($>\chi^2$)
Transect position	5.7743	4	0.21665
Aquaculture type	6.3637	1	**0.01165**
Treatment	57.8562	1	**<0.001**
Check time	330.5039	1	**<0.001**
Transect position × aquaculture type	6.9406	4	0.13906

4.1. Fish community

More fish were sighted in LL aquaculture and eelgrass habitats than in OB aquaculture habitat, as evidenced by the significant interaction between transect position and aquaculture type in our generalized linear model. This finding indicates that aquaculture method influenced the distribution of the observed fish species across the habitat transition, supporting our expectations of structural effects on nekton. To date, there have been few studies directly comparing LL and OB aquaculture. Hosack et al. (2006) found few differences in mobile nekton community composition or abundance in eelgrass, OB culture, or open unstructured habitats in Willapa Bay sampled with fyke nets that integrated catch over a 24 h period. With the exception of shiner perch *Cymatogaster aggergata*, which were more abundant in eelgrass habitat, a similar lack of habitat differences was reported by Dumbauld et al. (2015), who used actively towed trawl nets in order to capture juvenile salmonids. Pinnix et al. (2005) deployed shrimp trawls and fyke nets adjacent to off-bottom oyster culture,

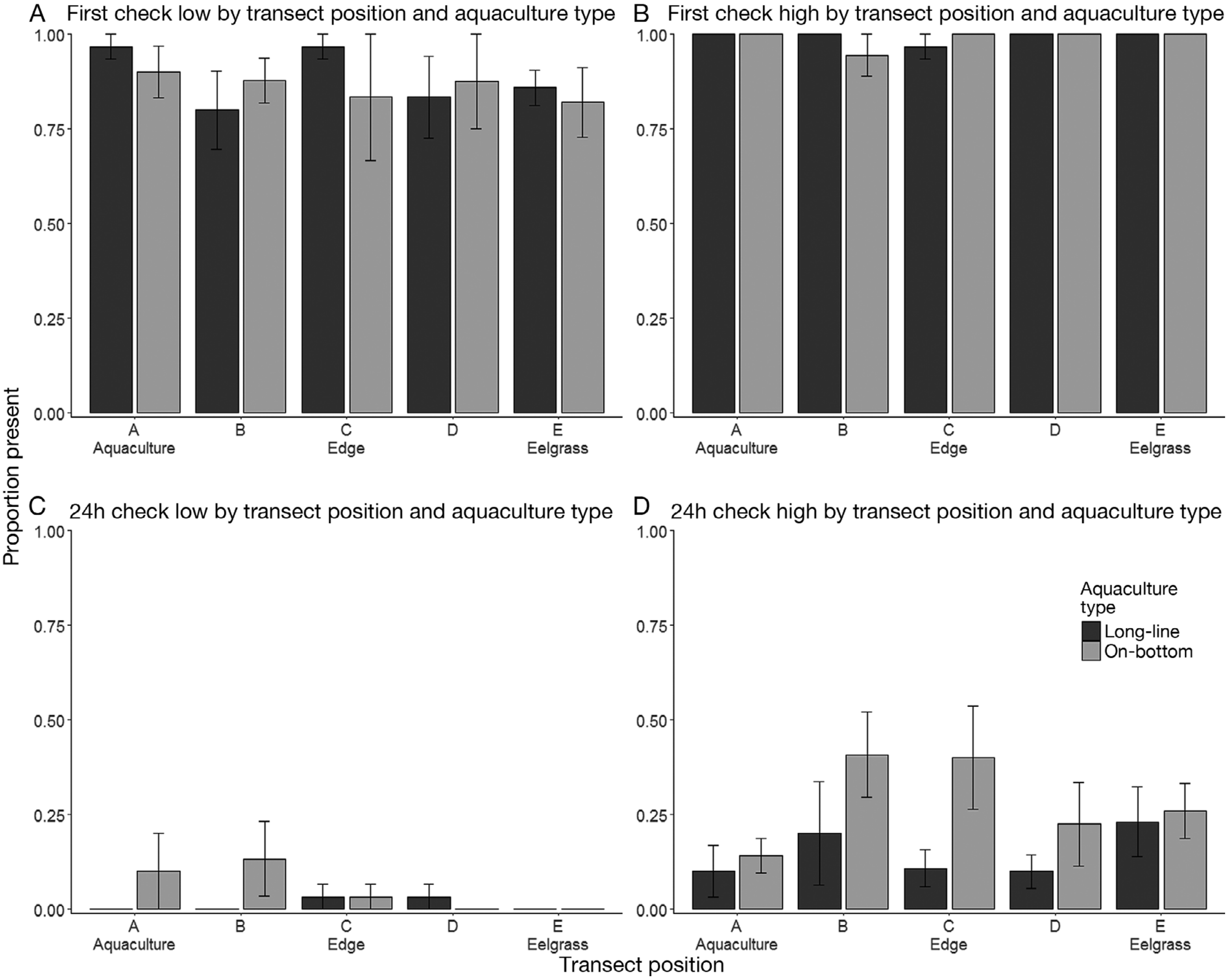

Fig. 6. Results of predation tethering unit arrays for both the (A,C) low treatment and (B,D) high treatment at the (A,B) first check and (C,D) 24 h check (N = 6). Error bars: ±1 SE

eelgrass, and open unvegetated mudflat habitats in Humboldt Bay, CA, and recorded higher species richness and higher catches adjacent to off-bottom culture habitat than either of the other 2 habitats. Off-bottom aquaculture also harbored a greater number of species than nearby eelgrass in an estuary in southern Rhode Island (DeAlteris et al. 2004). Although methods varied across these studies, trends in habitat use support the findings presented here.

The nekton community observed utilizing intertidal aquaculture and eelgrass habitats in our study was a subset of that documented as mesopredators in previous studies of these habitats in Willapa Bay and other estuaries along the US West Coast (Hosack et al. 2006, Hughes et al. 2014, Dumbauld et al. 2015, Gross et al. 2017). We caught fewer pelagic species and some of the less common taxa of nekton and attribute this primarily to differences in sampling gear and the total area sampled by our underwater video cameras. We used minnow traps to confirm the species seen in the video and provide a second data set to evaluate sampling discrepancies encountered when using video recording, like fish attraction to structures used to deploy cameras (Gross et al. 2018). While the species sampled were consistent, abundance of individual taxa sometimes differed between gear types. For example, the high catch of three-spined sticklebacks *Gasterosteus aculeatus* at one site (Tokeland) was corroborated by frequent sightings of this species in video from that site, but more saddleback gunnels *Pholis ornata*, a cryptic species, were caught in traps than sighted in video from the same locations. Additionally, while Dungeness crab *Metacarcinus magister* appeared most frequently in

OB culture habitat in both video and traps, as has been previously reported (reviewed in Dumbauld et al. 2009), we were less confident in these data for small crabs as they too have highly cryptic behavior in structured habitat. Thus, method considerations need to be carefully addressed when characterizing fish communities, particularly regarding management decisions.

While edge effects were not broadly apparent and congruous for both aquaculture methods, slightly fewer fish were observed at the edge. The lack of comprehensive statistical significance of edge effects could be due to the small sample size but perhaps also to the wide range of possible responses across species and time. Nonetheless, the visual trend in use of the observed species provides evidence that the habitat edge can influence species behavior. Edge sightings were significantly fewer than sightings in both the LL aquaculture and eelgrass interior habitats, suggesting that the edge provided less desirable habitat, potentially because of a higher predation risk or reduced food availability (Smith et al. 2011, Macreadie et al. 2012).

Responses of shiner perch and Pacific staghorn sculpins *Leptocottus armatus*, the 2 most sighted species in our study, further highlight potential driving mechanisms. Shiner perch were approximately equally abundant across the LL transect but were less abundant in OB aquaculture. Previous researchers have also documented higher abundance of shiner perch in eelgrass patches or on the edges of eelgrass beds relative to unvegetated sites (Dumbauld et al. 2015, Gross et al. 2017). In contrast, Pacific staghorn sculpins were found primarily in aquaculture habitats, especially LLs, and less frequently in interior eelgrass habitat. In Samish Bay (Washington, USA), Pacific staghorn sculpins were also sighted in underwater video more frequently in LL aquaculture and edge habitats than in eelgrass (Clarke 2017). The primary foraging behavior of each species may relate to contrasting use of edge habitats. Pacific staghorn sculpins have been recognized as avid predators of epibenthic invertebrates, juvenile fish, and insects, while shiner perch rely primarily on benthic polychaetes and other invertebrate mesograzers (Williams 1994, Troiano et al. 2013, McPeek et al. 2015, Whitney et al. 2017). Pacific staghorn sculpins make use of the refuge along the edge to capture prey, and shiner perch typically rely on the structure and food available in the interior of structured habitats. These differences in sightings may also relate to biases of the sampling method; shiner perch are often obvious swimming within the water column while Pacific staghorn sculpins swim along the bottom in a slow and cryptic manner and have previously been reported to be attracted to gear (Gross et al. 2018). Regardless, the marked differences in the distribution patterns of these 2 species highlight the complexities of edge effects between aquaculture and eelgrass.

4.2. Habitat structure

There was a clear increase in eelgrass habitat structure moving from both aquaculture habitats to eelgrass beds, as expected, given the sampling design, but the increase of eelgrass was less extreme on the LL transect than the OB transect (Fig. 3). This difference between aquaculture types is consistent with previous research that showed less impact of LL aquaculture on shoot density than OB culture, though both resulted in lower densities, and this difference was largely associated with mechanical harvest methods used in some OB culture areas (reviewed in Ferriss et al. 2019). Although we did not survey the growers to determine the most recent harvest event or method, all 3 OB culture beds we studied were previously classed as mechanical harvest (B. R. Dumbauld pers. obs). The LLs themselves also clearly provide vertical structure that contributes to the overall habitat complexity. DeAlteris et al. (2004) found that the emergent surface area ($cm^2\ m^{-2}$) of off-bottom aquaculture gear (racks) was comparable to that of eelgrass. Measurements of vertical structure provided by aquaculture gear would allow for a more complete understanding of the total structure that is potentially shaping the distribution of species across the habitat matrix.

The difference in eelgrass structure between aquaculture types also relates to the potential strength of edge effects between aquaculture and eelgrass beds. For both aquaculture types, the edge was defined based on where the aquaculture stopped; this did not always equate to the edge of the eelgrass bed. By this definition, the edge did not always represent a dramatic change in habitat. The type of edge (sharp vs. gradual transition) influences the magnitude of interactions shaping the distribution of marine mollusks across experimental landscapes, with edge effects being more pronounced when there was a sharper transition between aquaculture and adjacent habitat (Matias et al. 2013). The influence of the type of edge adds a layer of complexity to management of aquaculture and eelgrass, since the actual extent of eelgrass within an aquaculture bed changes over time

(Dumbauld & McCoy 2015). Further research could help to clarify how the sharpness of the edge affects habitat use.

4.3. Predation and resource availability

Results from our surveys using standardized PTUs were consistent with video sightings and provide evidence of greater levels of predation within the LL habitats than OB habitats for both treatments (Fig. 6). Increased predation intensity observed within the LL aquaculture is consistent with greater sightings of predatory fish (such as Pacific staghorn sculpins) and crabs in long-lines and suggests that the increased sightings were at least in part sightings of predators that consumed the bait. These patterns in bait consumption agree with those reported by Clarke (2017), who observed even higher loss of baits in Tillamook Bay, Oregon, but lower bait loss in Samish Bay, Washington, where she also observed fewer sculpins. Ruesink et al. (2019) reported a 70% loss of baits deployed in Willapa Bay for 24 h at the sediment surface during summer, with no difference in bait consumption between eelgrass and open mud habitats. Previous research suggests a potentially conflicting relationship between predation intensity and habitat structure. Some researchers have found that the presence of structure, but not the relative amount of structure, has an influence on prey survival (Mattila et al. 2008), while others have seen a clear decrease in predation risk with increased shoot density (Reynolds et al. 2018). This decrease is attributed to a reduction in detection and capture of mobile prey within more complex habitats. We saw an increase in predation with increasing habitat structure, suggesting that the complexity was not great enough to reduce detection of prey (i.e. bait).

Epiphytes on seagrass blades are known to be an important food source for epifauna or mesograzers within seagrass meadows (Jernakoff et al. 1996, Cullen-Unsworth & Unsworth 2013, Hayduk et al. 2019) and some of these mesograzers in turn are fed upon by mesopredators, including juvenile salmon. Quantification of epiphyte percent cover and load therefore provides an estimate of food availability among the habitat types. Values for epiphyte load in this study fell within the ranges of those found in other bays along the West Coast of the USA (Clarke 2017, Hayduk et al. 2019). Epiphyte load did not significantly vary across the transect or between aquaculture types in our study, suggesting that the presence of aquaculture did not affect epiphyte abundance on eelgrass. When considering total epiphyte biomass available at the landscape level (i.e. by multiplying epiphyte load by eelgrass shoot density), we did detect lower abundances in the aquaculture habitats due to the lower presence of eelgrass within these habitats. We did not, however, measure epiphytes present on the aquaculture gear or oysters themselves, which have been shown to be abundant elsewhere and would increase total food available to mesograzers (DeAlteris et al. 2004, Erbland & Ozbay 2008).

One of the advantages of using video as a sampling tool is the ability to make behavioral observations. We found that foraging behavior varied significantly by transect position, with less foraging occurring at the edge than in both interior habitats. In an Australian estuary, predators made use of edges between seagrass beds and open sand, while prey species were more common within the seagrass (Smith et al. 2011). These researchers found that predator presence along the edge may have discouraged foraging in this habitat. While observation of foraging in our study was limited (8% of all sightings), sightings of a known predator (Pacific staghorn sculpin) were low on the edge between aquaculture and eelgrass. Thus, we suggest that a combination of food availability and predation risk (perhaps by larger predators like birds) influenced by habitat structure shaped the decrease in foraging observed at this edge, but further research should be conducted to clarify this result.

5. CONCLUSIONS

This research sought to compare the use of habitat created by 2 different oyster aquaculture methods with that of eelgrass, which is an important natural habitat for fish and mobile invertebrates in a US Pacific coast estuary. We found a clear difference in fish use of off-bottom (LL) and OB oyster culture habitats, but edge effects were less distinct. The effect of structured habitat was species-specific, as had previously been observed for eelgrass, with 2 of the most abundant species, shiner perch *Cymatogaster aggergata* and Pacific staghorn sculpin *Leptocottus armatus* differing in their response.

Federal and state regulations currently prohibit the establishment of new aquaculture beds within native eelgrass and establish buffer zones around these eelgrass beds due to the recognized value of this structured habitat as a nursery for juvenile fish and invertebrates. For example, the US Army Corps of Engineers, with input from the NOAA National

Marine Fisheries Service, prohibits new aquaculture within a 16 horizontal foot buffer around eelgrass, while a buffer of 25–30 feet (7.6–9.1 m) is suggested under the Magnuson-Stevens Act (Pacific Fishery Management Council 2014, US Army Corps of Engineers 2017). While existing aquaculture is generally excluded from these regulations, the habitat provided by aquaculture itself is not valued, and this precautionary approach to expansion is based solely on autecology of eelgrass and value to other resources. Our results suggest that the type of aquaculture and structure it provides influences its functional value as estuarine habitat and that it is comparable to eelgrass for some resources. The type of aquaculture may also affect the breadth and strength of edge effects at larger scales and could inform concerns about eelgrass bed fragmentation and be a foundation for setting scientifically appropriate buffer widths between habitats. In addition, studies such as this could inform best management practices for shellfish aquaculture, which is increasingly being recognized as a beneficial use of US coastal waters (NOAA 2018). As the industry is expected to grow in the coming years, it is necessary to understand how that growth impacts estuarine function. Expanding the knowledge base concerning similarities between shellfish aquaculture and eelgrass will give managers tools to appropriately balance these 2 stakeholder interests.

Acknowledgements. This work was funded in part by 2 NOAA NMFS Saltonstall-Kennedy grants (2014/2015 NOAA-NMFS-FHQ-2015-2004246 and 2016/2017 NA16 NMF4270254), a 2016–2018 Oregon State University Agricultural Research Foundation grant to F.T., and project funds of the USDA Agricultural Research Service (CRIS Project 2072-63000-004-00D). Field and laboratory support were provided by D. Mercer, J. Minch, A. Blanchette, and G. Schwinge. Additional thanks to S. Heppell for support in developing the analysis.

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Editorial responsibility: Pablo Arechavala-Lopez, Esporles, Illes Balears, Spain

Submitted: September 30, 2019; Accepted: September 14, 2020
Proofs received from author(s): November 17, 2020

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Technical Report · October 2018

DOI: 10.25773/r01b-tg44

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Environmental and ecological benefits and impacts of oyster aquaculture: Addendum

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Recommended Citation

Kellogg, M. L., Turner, J., Dreyer, J. C., & Friedrichs, C. (2018) Environmental and ecological benefits and impacts of oyster aquaculture: Addendum. Virginia Institute of Marine Science, College of William and Mary. https://doi.org/10.25773/r01b-tg44

ENVIRONMENTAL AND ECOLOGICAL BENEFITS AND IMPACTS OF OYSTER AQUACULTURE: ADDENDUM

10/15/2018

Chesapeake Bay, Virginia, USA

An addendum to the final report to:
The Nature Conservancy

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Environmental and ecological benefits and impacts of oyster aquaculture: Addendum

CHESAPEAKE BAY, VIRGINIA, USA

Abstract

The data described in this addendum are provided to enhance the resolution and/or expand the temporal scope of the information already provided in the final report (Kellogg et al. 2018). High-resolution water quality transect data were collected at all four sites in Summer 2017, at White Stone (Windmill Point site) and Lynnhaven River in Fall 2017, and at White Stone (North Point site) in Spring 2018. During each sampling period, data were collected from multiple transects through and outside of each farm. Resulting data were detrended as needed based on temporal and salinity-related patterns found in data collected outside the farm footprint. Comparison of the resulting data from inside and outside the farm identified significant differences between water quality inside the farm footprint and outside for the majority of site x season combinations for all parameters. However, differences were consistently small enough to have no biologically significant impact, positive or negative, on farm-scale water quality.

Benthic macrofaunal communities inside and outside the farms were assessed at White Stone's Windmill Point site and at the Lynnhaven River site in Fall 2017 and White Stone's North Point site in Spring 2018. Data on species richness, macrofauna abundance, and macrofauna biomass were compared between samples taken inside the farm footprint and outside the farm footprint for all site x season combinations. These data were compared to data previously reported from Summer 2017 collected at all four aquaculture sites. Overall, patterns in species richness and macrofauna abundance were not consistent across seasons within site, across sites within seasons or within gear type. With the exception of one of the farm sites studied, there was a trend towards increased macrofauna biomass inside the footprint of aquaculture farms. This pattern is consistent with the assumption that food for benthic macrofauna at these sites is enhanced by oyster biodeposition. Overall, we found no biologically significant negative impacts on macrofaunal communities inside aquaculture farms and some evidence that suggests a possible positive impact on benthic macrofauna production.

Project Narrative

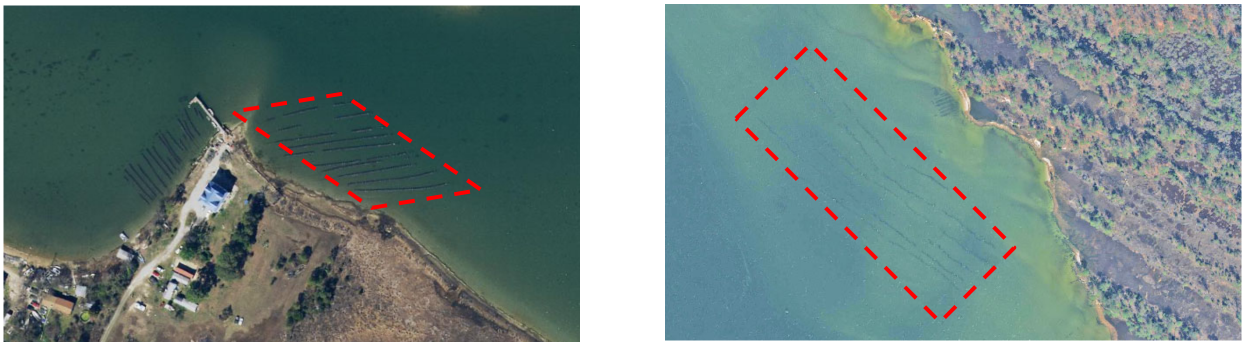
To better understand the environmental and ecological benefits and impacts of oyster aquaculture, we sampled water quality and benthic macrofaunal communities within oyster aquaculture sites and compared those data to data collected from the areas surrounding each farm. See Kellogg et al. (2018) for detailed information on farm locations and site characteristics.

Methods

Study sites: The same four sites were sampled during the same seasons as described in Kellogg et al. (2018). At each site, we delineated the footprint of the farm (hereafter "inside") based on a combination of GPS coordinates and aerial photography (Fig. 1). Data from inside the farm footprint were compared to data collected adjacent to but outside of the farms (hereafter "outside").

a) White Stone

b) Chapel Creek



c) Big Island

d) Lynnhaven

Fig. 1. Aerial images of aquaculture cages at each site showing differences in distribution of cages at each site. Lynnhaven aerial image has been edited to make cages more visible. Cages at Chapel Creek are difficult to distinguish from submerged aquatic vegetation in the vicinity of the cages. Dashed red lines indicate the approximate extent of the farm footprint studied at each site (Background images: VBMP).

Sampling periods: In Summer 2017, all four sites were sampled to compare water quality and macrofaunal community structure inside and outside the farms. In Fall 2017, additional samples of each type were taken at White Stone's Windmill Point site and at the Lynnhaven River site. In Winter 2017/2018, White Stone shifted production from the Windmill Point location previously sampled to a nearby location at North Point. Water quality and macrofauna community structure were assessed at White Stone's North Point site in Spring 2018.

Water quality transects: During each sampling period, data were collected from along transects that ran upstream, downstream and through the aquaculture farm parallel to aquaculture gear (Fig. 2). To increase the likelihood of detecting the influence of oyster aquaculture on water quality, data were collected from the upper portion of the water column at floating aquaculture sites and from the lower portion of the water column as sites utilizing bottom cages, with adjustments made as needed to avoid oyster cages. Along each transect, an acoustic Doppler current profiler (ADCP) was used to measure current velocity and a YSI 6600-series sonde was used to measure temperature, salinity, dissolved oxygen concentration, chlorophyll *a* fluorescence, pH and turbidity. All data were paired with location information from a GPS unit.

Benthic macrofaunal community: To assess benthic macrofaunal communities inside and outside the farm at each site, divers used hand cores to collect samples from a 62.2cm^2 area to a depth of ~10cm at all four sites in Summer 2017 and at White Stone and Lynnhaven River sites in Fall 2017. In Spring 2018, a petite ponar grab (216 cm^2 sample area) was used to collect macrofauna community samples from White Stone's North Point site. Samples were sieved immediately after collection and all material retained on a 1-mm mesh was fixed in Normalin for later analysis in the laboratory. In the laboratory, all organisms in samples were identified to the lowest practical taxon and counted. Because individual biomasses were small and abundances were generally low, organisms were pooled by major faunal groups within each sample prior to drying and weighing. All samples were dried to a constant weight at 60˚C. After dry weight data were collected, all samples were placed in a muffle furnace at 500˚C and burned to determine ash weight. Ash-free dry weight was determined by subtracting ash weights from dry weights.

Statistical analyses: Prior to statistical analyses, all water quality transect data were detrended as needed based upon data collected outside the farm footprint. This process consisted of regressing outside data against time of collection using a first, second or third order polynomial regression. If the regression was significant, the time trend was removed from the entire dataset (i.e. inside and outside data) using the regression function. Data were then regressed in the same manner against salinity and any significant salinity trends were removed. This approach results in means for the outside data that approach zero and means for the inside data that are positive if the measured parameter is higher inside the farm and negative if it is lower inside the farm.

The effect of farms on water quality and benthic community structure was determined using one-way ANOVAs with two levels (inside farm and outside farm) to determine significant differences between means for all macrofauna community parameters and detrended means for all water quality parameters. Data that violated ANOVA assumptions of normality and/or equal variance were transformed as needed to meet these assumptions. Some of the water quality datasets that violated ANOVA assumptions of normality and/or equal variance were resistant to transformation. In these cases, we assumed that ANOVA were robust to these violations, an assumption justified in part by the large number of samples included in analyses. For all tests, p-values ≤ 0.05 were considered statistically significant.

Results

Water quality transects: Significant differences in water quality inside and outside the farm footprint were common for all parameters during all sampling periods. Only the Chapel Creek site showed no significant differences between water quality inside and outside the farm for all parameters. A finding that is likely related to the small size of the farm footprint at this site and resulting in relatively low sample numbers compared to other sites. Although the farms frequently had impacts on water quality, the scale of these impacts was consistently small enough to be biologically insignificant, leading to the conclusion that these farms have minimal positive or negative impact on water quality.

Current speed: Flow was significantly reduced in four of the seven sets of samples and was not significantly different in the other three sets. Big Island, the site with the highest density of cages, had the greatest effect on flow, reducing it by more than 4 cm sec^{-1}. Flow was also reduced by ~2-3 cm sec^{-1} at White Stone's Windmill Point site (sampled in summer and fall) and at the Lynnhaven site in fall. Chapel Creek, the site with the smallest footprint, had no detectable effect on flow.

Table 1. Effect of aquaculture farm on current speeds measured inside and outside the farm footprint. Refer to text for methods used to detrend data. Significant differences between detrended means are indicated by "<" or ">" depending upon whether means were lower or higher, respectively, inside the footprint of the farm. Means that are not siginifcantly different are indicated by "≈".

Farm	**Season**	**Samples (#)**		**Current Speed (cm/sec)**		**Detrended Current Speed (Mean ± SD)**		
		Inside	*Outside*	*Inside*	*Outside*	*Inside*	*Sig.*	*Outside*
Big Island	Summer	370	1121	10.8	10.9	-4.25 ± 10.68	<	0.00 ± 6.04
Chapel Creek	Summer	57	693	9.4	8.0	1.34 ± 6.14	≈	0.00 ± 5.29
Lynnhaven	Summer	797	1893	11.3	11.8	0.49 ± 6.33	≈	0.00 ± 5.95
	Fall	338	996	11.9	12.2	-2.29 ± 6.92	<	0.00 ± 6.13
White Stone	Spring	788	1062	16.2	15.9	-0.11 ± 9.49	≈	0.00 ± 9.42
	Summer	859	1533	16.8	18.4	-2.14 ± 7.33	<	0.00 ± 7.83
	Fall	657	1948	13.4	16.7	-3.11 ± 7.64	<	0.00 ± 9.08

Dissolved oxygen: Aquaculture farms had a very small but significant positive effect on dissolved oxygen in five of the seven datasets (Table 2). Data indicate that farm-scale water quality at these sites is not negatively impacted by oyster respiration. It is important to note that all data were collected during daylight hours. Visual observations at White Stone's North Point site suggest that the slight increase in dissolved oxygen within farms may be attributable to benthic microagal or macroalgal growth on aquaculture gear, leading to increased rates of photosynthesis during daylight hours resulting in increased oxygen concentration. At night, when photosynthesis shuts down, it is possible that different patterns in oxygen concentration would be observed.

Table 2. Effect of aquaculture farm on dissolved oxygen measured inside and outside the farm footprint. Refer to text for methods used to detrend data. Significant differences between detrended means are indicated by "<" or ">" depending upon whether means were lower or higher, respectively, inside the footprint of the farm. Means that are not siginifcantly different are indicated by "≈".

Farm	**Season**	**Samples (#)**		**Dissolved Oxygen (mg L^{-1})**		**Detrended Dissolved Oxygen (Mean ± SD)**		
		Inside	*Outside*	*Inside*	*Outside*	*Inside*	*Sig.*	*Outside*
Big Island	Summer	159	778	7.4	7.3	0.01 ± 0.10	≈	0.00 ± 0.11
Chapel Creek	Summer	31	383	7.2	7.1	0.00 ± 0.03	≈	0.00 ± 0.05
Lynnhaven	Summer	523	1259	6.1	5.9	0.08 ± 0.09	>	0.00 ± 0.07
	Fall	245	732	6.1	6.1	0.07 ± 0.11	>	0.00 ± 0.12
White Stone	Spring	549	763	9.7	9.7	0.02 ± 0.03	>	0.00 ± 0.05
	Summer	564	1109	8.4	8.4	0.01 ± 0.04	>	0.00 ± 0.06
	Fall	448	1526	7.9	7.9	0.02 ± 0.04	>	0.00 ± 0.07

pH: Effects on pH were significant but extremely small with a maximum difference between means of 0.023 (Table 3). In five of the seven datasets, pH was slightly higher inside the farm. Because increases photosynthesis lead to increases in pH, these findings are generally consistent with dissolved oxygen results. As for dissolved oxygen, all samples were taken during daylight hours making it possible that different patterns in pH would be observed at night.

Table 3. Effect of aquaculture farm on pH measured inside and outside the farm footprint. Refer to text for methods used to detrend data. Significant differences between detrended means are indicated by "<" or ">" depending upon whether means were lower or higher, respectively, inside the footprint of the farm. Means that are not siginifcantly different are indicated by "≈".

Farm	**Season**	**Samples (#)**		**pH**		**Detrended pH (Mean ± SD)**		
		Inside	*Outside*	*Inside*	*Outside*	*Inside*	*Sig.*	*Outside*
Big Island	Summer	159	778	7.9	7.9	-0.023 ± 0.030	<	0.000 ± 0.022
Chapel Creek	Summer	31	383	8.1	8.1	0.004 ± 0.005	>	0.000 ± 0.009
Lynnhaven	Summer	523	1259	7.9	7.8	0.011 ± 0.011	>	0.000 ± 0.011
	Fall	245	732	7.6	7.6	0.004 ± 0.011	>	0.000 ± 0.016
White Stone	Spring	549	763	8.4	8.4	-0.005 ± 0.019	<	0.000 ± 0.024
	Summer	564	1109	8.1	8.1	0.002 ± 0.009	>	0.000 ± 0.011
	Fall	448	1526	8.0	8.0	0.000 ± 0.005	>	0.000 ± 0.011

Chlorophyll: Aquaculture farms had significant but very small effects on chlorophyll with no consistent direction in the effect (Table 4). Three of the datasets indicate enhanced chlorophyll concentrations, three indicate reduced concentrations, and, again, there was no significant effect of the farm at Chapel Creek. Through their feeding activities, oysters are expected to reduce chlorophyll concentrations in the water column because they filter and consume phytoplankton. Although the production of oysters for harvest at these sites makes it clear that oysters are consuming phytoplankton, the scale of that consumption appears to be small enough that it has minimal impacts on water quality at the farm scale. Note that the method used to assess farm-scale chlorophyll concentrations measures only the amount of chlorophyll suspended in the water column and could not account for the chlorophyll in algae attached to aquaculture gear. This incomplete accounting of chlorophyll at the farm scale may partially explain the lack of a consistent pattern between dissolved oxygen data and chlorophyll data.

Table 4. Effect of aquaculture farm on chlorophyll measured inside and outside the farm footprint. Refer to text for methods used to detrend data. Significant differences between detrended means are indicated by "<" or ">" depending upon whether means were lower or higher, respectively, inside the footprint of the farm. Means that are not siginifcantly different are indicated by "≈".

Farm	**Season**	**Samples (#)**		**Chlorophyll ($\mu g\ L^{-1}$)**		**Detrended Chlorophyll (Mean ± SD)**		
		Inside	*Outside*	*Inside*	*Outside*	*Inside*	*Sig.*	*Outside*
Big Island	Summer	159	778	9.2	9.6	-0.19 ± 0.77	<	0.00 ± 0.73
Chapel Creek	Summer	31	383	3.6	3.6	-0.11 ± 0.58	≈	0.00 ± 0.57
Lynnhaven	Summer	523	1259	12.6	12.6	-0.11 ± 0.77	<	0.00 ± 0.57
	Fall	245	732	5.4	5.1	0.18 ± 0.53	>	0.00 ± 0.54
White Stone	Spring	549	763	2.4	2.5	-0.04 ± 0.29	<	0.00 ± 0.42
	Summer	564	1109	1.8	1.8	0.12 ± 0.93	>	0.00 ± 0.54
	Fall	448	1526	2.9	2.6	0.38 ± 1.52	>	0.00 ± 0.88

Turbidity: Turbidity is a measure of the amount of light scattered by particles in the water column. As oysters filter feed, they consume both sediments and phytoplankton. Sediments are repackaged into pseudofeces that are larger and have a greater sinking velocity than the sediments prior to filtration. These larger particles scatter less light leading to the expectation that oyster feeding activities reduce turbidity via both direct consumption of phytoplankton and repackaging of sediments. Turbidity was significantly lower inside the farm for five of the seven datasets (Table 5). Again, no significant difference was found at Chapel Creek, likely due to the small footprint of the farm. As for other parameters, the magnitude of differences between inside and outside the farm are not large enough to be biologically meaningful.

Table 5. Effect of aquaculture farm on turbidity measured inside and outside the farm footprint. Refer to text for methods used to detrend data. Significant differences between detrended means are indicated by "<" or ">" depending upon whether means were lower or higher, respectively, inside the footprint of the farm. Means that are not siginifcantly different are indicated by "≈".

Farm	**Season**	**Samples (#)**		**Turbidity (NTU)**		**Detrended Turbidity (Mean ± SD)**		
		Inside	*Outside*	*Inside*	*Outside*	*Inside*	*Sig.*	*Outside*
Big Island	Summer	159	778	5.5	6.3	-0.59 ± 0.30	<	0.00 ± 0.41
Chapel Creek	Summer	31	383	1.7	1.7	0.04 ± 0.05	≈	0.00 ± 0.13
Lynnhaven	Summer	523	1259	9.7	10.0	-0.63 ± 0.67	<	0.00 ± 0.64
	Fall	245	732	4.7	4.9	-0.43 ± 0.22	<	0.00 ± 0.36
White Stone	Spring	549	763	0.3	0.3	-0.10 ± 0.17	<	0.00 ± 0.19
	Summer	564	1109	1.8	1.7	0.05 ± 0.22	>	0.00 ± 0.23
	Fall	448	1526	1.2	1.3	-0.08 ± 0.10	<	0.00 ± 0.43

Temperature: Although significant effects on temperature were identified for all sites except Chapel Creek, the scale of the impact was extremely small with a maximum difference of 0.09 ˚C (Table 6). The two sites with floating aquaculture gear consistently reduced water temperatures. One possible cause for reduced temperatures at these sites is shading of the water column by the floating aquaculture gear. Observation of the greatest impact at Big Island, the site with the highest gear density, is consistent with this hypothesis. Again, the small spatial extent and relatively small sample size of the Chapel Creek farm may partially explain its lack of significant impact.

Table 6. Effect of aquaculture farm on temperature measured inside and outside the farm footprint. Refer to text for methods used to detrend data. Significant differences between detrended means are indicated by "<" or ">" depending upon whether means were lower or higher, respectively, inside the footprint of the farm. Means that are not siginifcantly different are indicated by "≈".

Farm	**Season**	**Samples (#)**		**Temperature (°C)**		**Detrended Temperature (Mean ± SD)**		
		Inside	*Outside*	*Inside*	*Outside*	*Inside*	*Sig.*	*Outside*
Big Island	Summer	159	778	25.1	25.2	-0.09 ± 0.10	<	0.00 ± 0.09
Chapel Creek	Summer	31	383	24.6	24.5	0.01 ± 0.02	≈	0.00 ± 0.02
Lynnhaven	Summer	523	1259	29.3	29.3	0.03 ± 0.02	>	0.00 ± 0.04
	Fall	245	732	22.2	22.4	-0.08 ± 0.16	<	0.00 ± 0.13
White Stone	Spring	549	763	18.8	18.8	-0.08 ± 0.07	<	0.00 ± 0.07
	Summer	564	1109	24.1	24.3	-0.04 ± 0.04	<	0.00 ± 0.06
	Fall	448	1526	19.2	19.3	-0.02 ± 0.08	<	0.00 ± 0.16

Benthic macrofaunal community: Data from summer 2017 have been reported previously but we include them again here to place them in the context of observed patterns for other seasons. Although significant differences were occasionally found between the benthic macrofaunal communities inside and outside the farm footprint at some sites during some seasons, there was no consistent effect, positive or negative, across sites in terms of species richness (Fig. 3) or macrofauna abundance (Fig. 4 and 5). There was a trend towards enhanced macrofauna biomass inside farms (Fig. 6).

Of the macrofauna community characteristics examined, species richness was the most consistent across sites and seasons (Fig. 3). Only two of the seven site x season datasets show significant differences between species richness inside and outside of the farm footprint, with White Stone's Windmill point site showing reduced species richness inside the farm footprint in summer and Lynnhaven showing increased species richness inside the farm footprint in summer. Of the other five site x season combinations, four showed a tendency towards higher species richness inside the farm footprint. Trends were not consistent across seasons within site, across sites within seasons or within gear type.

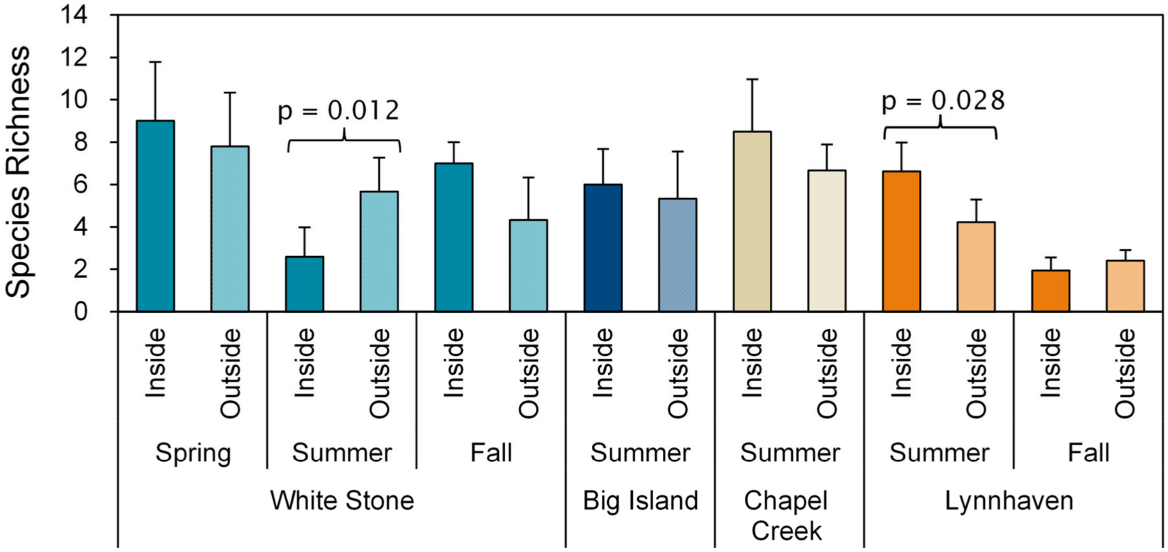


Fig. 3. Mean species richness for all sites and sampling periods. Error bars represent ± 1 standard deviation. P-values are given for site x treatment combinations in which there was a significant difference between areas inside and outside the farm footprint.

Macrofauna abundance showed greater variation than other macrofauna community characteristics. By far, the most organisms were found at White Stone's North Point site in Spring 2018 (Fig. 4). This pattern was driven almost entirely by high abundances of the amethyst gem clam, *Gemma gemma*. During this sampling period, the mean abundance of organisms inside the footprint of the aquaculture site was significantly higher than outside the footprint.

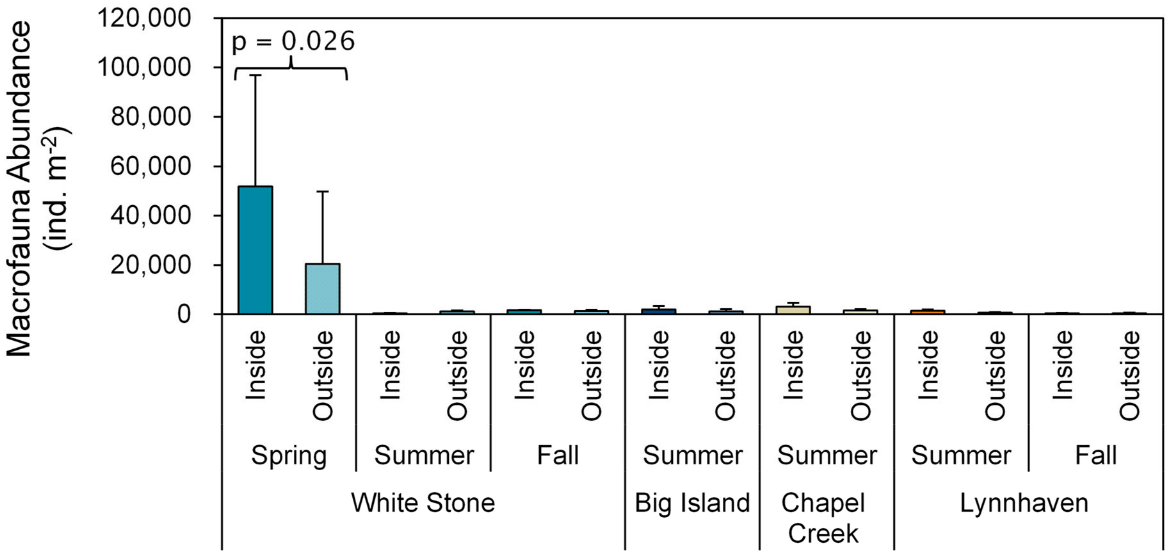


Fig. 4. Mean macrofauna abundance for all sites and sampling periods. Error bars represent ± 1 standard deviation. P-value is given only for White Stone spring samples. Other significant p-values are shown in Figure 5.

For the other six site x sampling period datasets (Fig. 5), significant differences between mean organism abundance inside and outside the farm footprint were observed three times, all during summer months. Mean abundances were higher inside the farm footprint at both Chapel Creek and Lynnhaven and significantly lower at White Stone's Windmill Point site. Of the other three site x season combinations, two showed

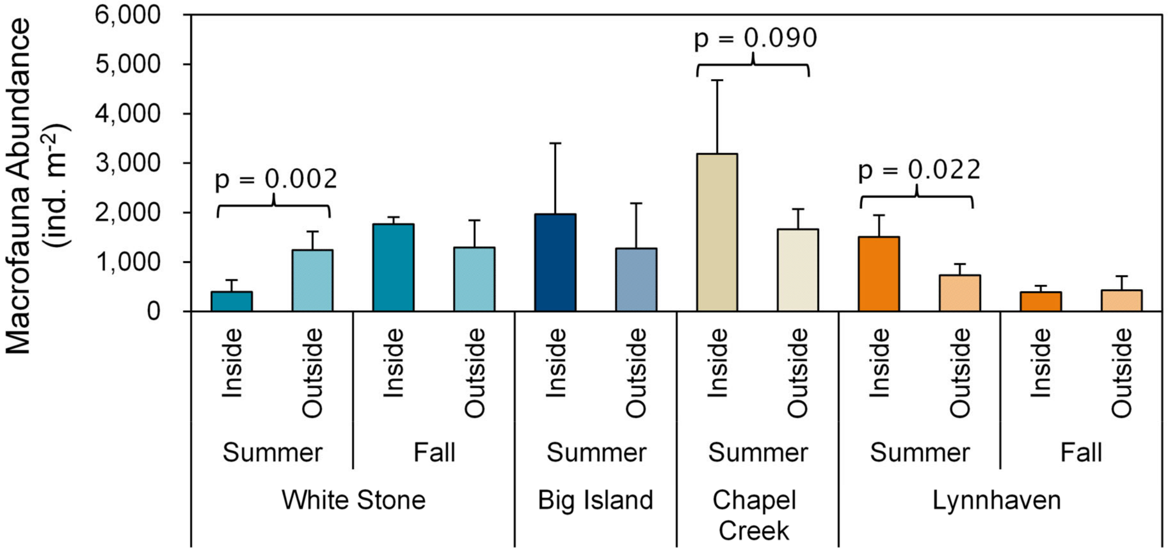


Fig. 5. Mean macrofauna abundance for all sites and sampling periods except White Stone spring samples. Error bars represent ± 1 standard deviation. P-values are given for site x treatment combinations in which there was a significant difference between areas inside and outside the farm footprint.

a tendency towards higher macrofauna abundance inside the farm footprint. Overall, trends in macrofauna abundance were not consistent across seasons within site, across sites within seasons or within gear type.

Mean macrofauna biomass only differed significantly between inside and outside the farm at White Stone's North Point site in spring where higher biomass was found within the footprint of the farm. This pattern is similar to that seen for macrofauna abundance and is again largely attributable to large populations *Gemma gemma* at the site. Of the other six site x season combinations, four show a tendency towards higher biomass inside the farm. The two that do not follow this pattern are the samples collected at White Stone's Windmill Point site. Observations made at the time of sampling suggest that the tendency towards slightly lower biomass inside the farm at Windmill Point is unlikely to be the result of negative impacts associated with enhanced organic deposition. The tendency towards higher biomass inside the farm footprint observed for the majority of season x site combinations is the most consistent pattern observed in macrofauna community structure during these studies and is consistent with the expectation that oyster biodeposits enhance the supply of food available to benthic organisms at these sites. This tendency towards increased benthic biomass inside the farms even during summer months suggests that organic loading of the sediments is not great enough to result in negative impacts on benthic habitat quality.

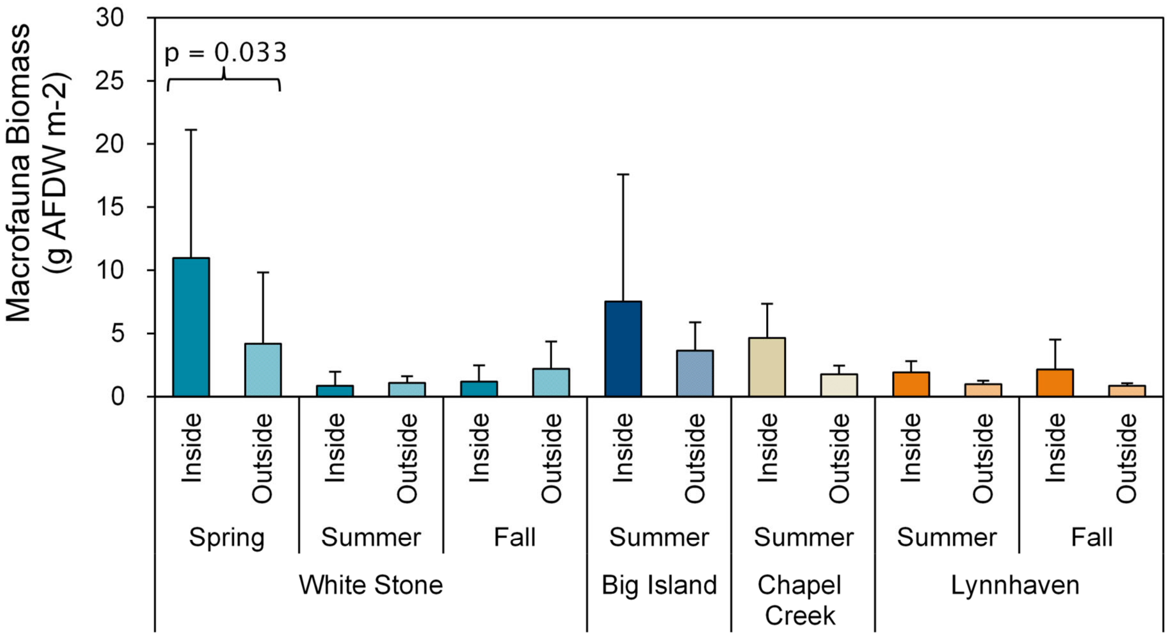


Fig. 6. Mean macrofauna biomass for all sites and sampling periods. Error bars represent ± 1 standard deviation. P-values are given for site x treatment combinations in which there was a significant difference between areas inside and outside the farm footprint.

Conclusions

- After studying a range of gear types, locations, and aquaculture farm scales along the western shore of Chesapeake Bay in Virginia, we found no evidence of biologically significant negative impacts on benthic macrofauna community structure or water quality.
- Our approach to measuring water quality differences between aquaculture farms and the surrounding area followed by detrending of data based on time and salinity allowed us to detect very small differences in water quality between waters within the farm footprint and areas outside the farm footprint for farms ≥1.35 acres. We believe this approach is likely to be useful for future studies of water quality impacts for farms of medium to large spatial scales.
- The failure to detect any significant water quality differences at Chapel Creek despite high oyster biomass density suggests that the effects of farms at this spatial scale (≤ 0.28 acres) are difficult to detect against background levels of variation in water quality parameters using the transect approach.
- Significant differences between water quality inside the farm footprint and outside were detected for the majority of site x season combinations for all parameters, but differences were consistently small enough to be biologically insignificant.
- Trends in species richness and macrofauna abundance were not consistent across seasons within site, across sites within seasons or within gear type.
- With the exception of one of the farm sites studied, there was a trend towards increased macrofauna biomass inside the footprint of aquaculture farms. This pattern is consistent with the assumption that food for benthic macrofauna at these sites is enhanced by oyster biodeposition. The tendency towards higher biomass inside the farm footprint during summer at three of the four sites studied also suggests that biodeposition rates are not high enough to result in degradation of benthic habitats at these sites.

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Aquaculture
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Impact of suspended and off-bottom Eastern oyster culture on the benthic environment in eastern Canada

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Received 21 September 2004; received in revised form 28 November 2005; accepted 30 November 2005

Abstract

The impact of Eastern oyster culture (*Crassostrea virginica*) on the benthic environment at a shellfish farm in New Brunswick, Canada, was assessed using recommended methods for routine environmental monitoring, specifically measurements of sediment redox and sulfide levels. Maximum culture density was equivalent to 4000 oyster bags per hectare, or a final oyster biomass of 8 kg m^{-2}. Two culture sites, one with floating bags and one with oyster tables, as well as two reference sites were monitored over 17 months (June 2002–October 2003). Seasonal variations in sediment redox and sulfide levels were observed, but no significant differences were detected between the culture and the reference sites. Biodeposition associated with the oyster biomass contributed to increased sedimentation rates of organic matter at the oyster table site, but there was no indication of organic enrichment in the sediment. Macrofauna biomass, abundance and number of species were higher at the oyster table site than at the other sites in September 2002, but values were similar for all sites in September 2003. In this region of eastern Canada, the bays are typically shallow and the upper layers of the sediment are frequently subjected to re-suspension by wave activity and physical erosion by winter ice. Given these highly dynamic conditions and the relatively low stocking densities per hectare, we would argue that the potential impact of oyster culture on the environment should be assessed on the basis of parameters other than sediment redox and sulfide levels.

Keywords: Shellfish farming; Oyster culture; Environmental impact; Sulfide; Redox; Sedimentation rates

1. Introduction

One of the earliest studies aimed at evaluating the impact of shellfish culture on the benthic environment was conducted in a Swedish inlet where sulfide levels in the sediment under the mussel lines were observed to be 100 times higher than at the reference site (Dahlback and Gunnarsson, 1981). Examination of the benthic macrofauna suggested that these high sulfide levels and associated anoxia resulted in a transition towards communities dominated by opportunistic polychaetes (Mattson and Linden, 1983). Similar biochemical and biological changes were documented in other areas of intensive mussel longline culture such as New Zealand (Kaspar et al., 1985), northern Russia (Chivilev and Ivanov, 1997) and Ireland (Chamberlain et al., 2001). In contrast, research conducted in eastern Canada at a site with relatively low mussel stocking densities suggested

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doi:10.1016/j.aquaculture.2005.11.054

no negative impact on the benthic community, with the exception of a partial shift toward anaerobiosis directly under the mussel lines (Hatcher et al., 1994; Grant et al., 1995). Researchers in Carteau, France concluded that increased biodeposition associated with mussel culture did not have a detrimental effect on the ecosystem, although there was a localized increase in bacterial activity and the flow of nutrients (Baudinet et al., 1990; Grenz et al., 1990). Likewise, a review of mussel culture practices in Chile pointed to minimal effects on the benthic environment (Buschmann et al., 1996).

Studies on the environmental impact of intensive mussel raft culture have also yielded mixed results. In South Africa, the accumulation of refractory organic matter under mussel rafts was shown to have a substantial negative impact on the macrobenthic community, and only marginal recovery was observed 4 years after their removal (Stenton-Dozey et al., 1999, 2001). In the Rias of northwestern Spain, heavy organic enrichment associated with biodeposition was also linked to significant changes in the infaunal benthic community (Tenore et al., 1982). However, other studies of the same ecosystem suggested that certain types of macrofauna, such as crabs and demersal fishes, benefited from the additional food supply associated with mussel fall-off as well as from the increase in the population of deposit-feeding prey organisms (Iglesias, 1981; Romero et al., 1982; Lopez-Jamar et al., 1984; Gonzalez-Gurriaran, 1986; Freire et al., 1990).

Most studies on the environmental impact of intertidal oyster culture originate from intensively cultured areas on the Atlantic coast of France (e.g. the Marennes-Oléron Basin). Early studies indicated that the deployment of oyster tables in physically dynamic intertidal environments could obstruct current flow patterns leading to increased sedimentation rates and decreased mechanical erosion (Ottman and Sornin, 1985). It was argued that the production of hydrogen sulfide associated with these high sedimentation rates could pose a threat to oysters reared in small or poorly circulating bays (Mariojouls and Sornin, 1987). Results of other studies on intertidal oyster culture, however, demonstrated minimal longterm environmental impacts. Martin et al. (1991) measured a significant accumulation of biodeposits under oyster tables, but no difference between the experimental and the reference sediments was detected 2 months after removal of the oysters. Similarly, another study found no significant release of nutrients from oysters cultured on intertidal tables and concluded that most of the biodeposition was flushed away and mineralization occurred elsewhere (Smaal and Zurburg, 1997). These variable results illustrate the importance of the interaction between the particular grow-out structure, the intensity of culture and the local environmental characteristics (Castel et al., 1989).

A series of studies on the impact of intensive subtidal oyster culture in the Thau lagoon on the French Mediterranean coast indicated significant changes in sediment biochemistry and localized nutrient cycling. Enhanced microphyte activity under the oyster tables resulted in consistently higher chlorophyll levels and ammonia release rates than in reference sediments (Barranguet et al., 1994). Under conditions of high biodeposition, this microphyte activity was insufficient to meet the oxygen demand which led to localized oxygen depletion, sulfide accumulation and extensive macrofaunal mortality. The composition of the surface sediments and the associated benthic macrophyte community in various areas of the lagoon was linked to the intensity of shellfish culture; specifically, in areas with higher levels of biodeposition, there was a shift away from the original *Zostera* community towards more opportunistic and nitrophilous species such as *Ulva* and *Gracilaria* (De Casabianca et al., 1997). On the other hand, researchers also noted that the uptake of particulate matter by the oysters had increased water clarity in the lagoon to the extent that seagrasses such as *Zostera* could extend their range down to 5 m (Deslous-Paoli et al., 1998).

Recent environmental assessment studies of intertidal and subtidal oyster culture in Tasmania did not indicate any negative impacts on sediment biochemistry or macrofauna (Thorne, 1998; Crawford et al., 2003). It would appear that the environmental impacts of shellfish culture vary depending on the scale of culture, the culture method and the physical conditions. The existence of this variability points to the importance of conducting localized environmental assessments. In eastern Canada, two main techniques are currently used to grow oysters: floating oyster bags attached to longlines, and oyster bags deployed on off-bottom oyster tables. Because of the severe ice conditions and the shallow nature of the grow-out sites, the gear is typically moved onto the lease in early May after ice-break-up, and removed in early November with the onset of winter conditions. The oyster inventory is overwintered in deep water and then re-deployed at the same general location on the lease site the following spring. In effect, these non-permanent oyster culture practices constitute a seasonal disturbance factor which, similar to major winter storms, may or may not have long-term consequences for the health of the ecosystem. The purpose of the following study was to evaluate the impact of these two oyster culture methods on the sediment biochemistry, sediment deposition rates and the benthic community over a 17-month period.

2. Materials and methods

2.1. Location of study

The study was undertaken in South St-Simon Bay, located near the town of Shippagan, New Brunswick, Canada (47°43.398′ N 64°46.018′ W). The system is a large shallow inlet connected to the Bay of Chaleurs, with extensive eelgrass coverage, a mean low tide depth of 0.6 m, and ice coverage from November to April (Fig. 1). The sampling area was a 35-ha oyster lease which had been used for the bottom culture of oysters from 1982 until 1997 when the company switched to floating bag and oyster table culture techniques. Two experimental culture sites were selected within the lease area, one in the zone where the floating bags were anchored (Floating Bag) and the other in the zone where the oyster tables were deployed (Table). Culture density at the Floating Bag site was 2000 bags/ha (4 kg oyster biomass m^{-2}) compared to 4000 bags/ha (8 kg oyster biomass m^{-2}) at the Table site. Two reference sites were also selected, one located 100 m northwest of the culture activity (Reference North) and the other 100 m to the southwest (Reference South). This distance between the culture and reference sites was consistent with guidelines specified in other similar studies (Grant et al., 1995; Crawford et al., 2003). GPS coordinates were recorded for each site to facilitate their re-location and sampling under the winter ice. Note that the Floating Bag site and the two Reference sites were located in the upper subtidal zone with thick eelgrass coverage whereas the Table site was in the lower intertidal zone with relatively sparse eelgrass coverage.

2.2. Sediment biochemistry

Two core samples (5 cm in diameter × 15 cm deep) were obtained at low tide from each of the culture and the reference sites every 6 weeks from late June to November 2002, through the ice in February and April 2003, and again every 6 weeks from May to October 2003. During core collection water depths at the Table site were typically 0.5-m deep. The cores were transferred to a 4 °C cold room at the Shippagan Marine Center and processed within 12 h of collection. Redox and sulfide levels were assessed according to the procedures described in Hargrave et al. (1995) and Wildish et al. (1999).

Redox potential in the core samples was measured at 2-cm intervals using an Orion (9678BN) combination redox (platinum) electrode connected to a Hanna pH/mV meter. The electrode was calibrated using Zobell's ferro/ferricyanide standard solutions before and after sampling each core (Zobell, 1946). The probe was allowed to equilibrate for at least 30 s before taking each reading. Sulfide (S^{2-}) was measured using an Orion (9416BN) combination silver/sulfide electrode connected to an Accumet 1003 specific ion meter. Sediment samples (5 ml) were extracted from the core at

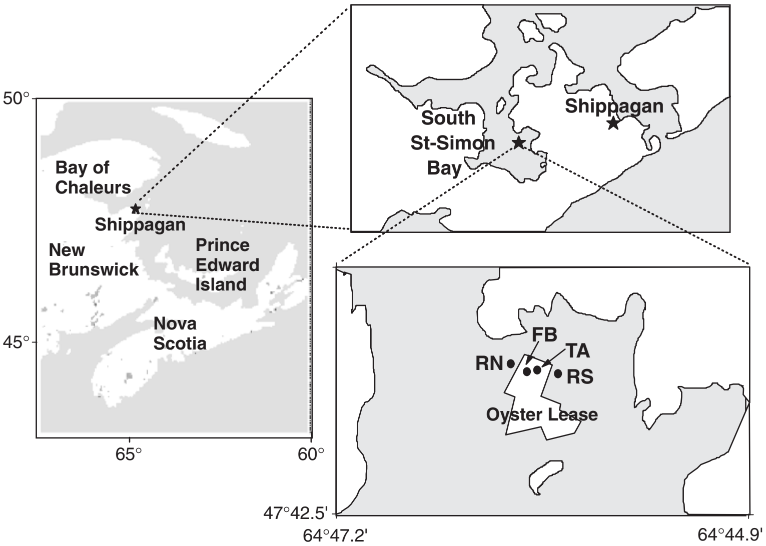

Fig. 1. Location of the oyster culture lease in South St-Simon Bay on the northeast coast of New Brunswick, Canada. The positions of the two reference sites (RN=Reference North, RS=Reference South) and the two experimental sites (FB=Floating Bags, TA=Tables) are indicated.

2-cm intervals using a modified 10-ml syringe with the funnel tip removed. Each sample was immediately mixed with an equal volume of freshly prepared sulfide anti-oxidant buffer (SAOB) solution. The sulfide electrode was calibrated using freshly prepared standard sodium sulfide solutions ($Na_2S \cdot 9H_2O$—1000, 100, 10 μM). Sulfide levels were recorded when the reading stabilised (i.e. ceased increasing), or approximately 10–15 min after the sulfide anti-oxidant buffer was added.

Additional samples from each 2-cm layer were transferred to pre-weighed scintillation vials for the determination of water and organic content in the sediment. The samples were dried for 48 h at 60 °C and re-weighed to obtain estimates of water content. The contents of each vial were pulverized and subsamples were then removed, weighed and ashed at 500 °C for 24 h to determine the organic content.

2.3. Sediment deposition

Sediment traps were deployed weekly at each sampling location in the summer 2002 and 2003. At the culture sites, one trap was deployed directly beneath the tables or the floating bags and the other between the rearing structures. Each trap consisted of two PVC cylinders (5 cm in diameter × 30 cm high) with a cap on the bottom and a plastic baffle fitted into the top. Each cylinder was secured with bungee cord in one corner of a plastic perforated tray weighted down with a brick. When the traps were retrieved, each replicate cylinder was swirled vigorously and the contents were transferred to an individual sampling bottle. After shaking each sampling bottle to resuspend the sediment, two replicate subsamples of the sediment slurry were collected on pre-weighed ashed GF/C filters, rinsed with 10-ml ammonium formate, and dried for 48 h at 60 °C. After weighing to determine total sediment weight, the samples were ashed at 450 °C for 4–6 h and re-weighed to determine sediment organic content.

2.4. Water column characteristics

At each sampling site, approximately 4 L of seawater were collected from 10 cm below the surface for measurements of chlorophyll and suspended particulate matter (biweekly in 2002, weekly in 2003). Two subsamples (2 × 1500 ml) were processed for TPM (total particulate matter) and POM (particulate organic matter) as described above for the sediment samples. Subsamples for chlorophyll (2 × 200 ml) were filtered (Whatman GF/C filters) and frozen immediately for subsequent analysis. Frozen filters were transferred to scintillation vials containing 10 ml 90% acetone and left for 24 h in the freezer. Chlorophyll levels (corrected for phaeophytin) were determined using a Turner fluorometer.

2.5. Benthic macrofauna

Two sediment samples (20 × 20 × 20 cm) were collected from each sampling site for assessment of macrofauna community composition in September 2002 and 2003. The sediment was sieved to collect all organisms >0.5 mm. After sorting and determining the wet weight of each taxon, the organisms were preserved in 10% buffered formalin for 2 weeks and then transferred to ethanol for long-term storage. Identification to species level was carried out where possible.

2.6. Data analysis

The main objective of this study was to evaluate whether any differences in the sediment parameters could be detected between the Reference and the Culture sites over the course of the study. For this reason, Time was declared a random factor and the repeated sampling using two cores at each location over time provided a robust comparison of the locations. In addition, all data points collected from each core (5 samples) were used in the statistical analyses. Sediment biochemistry data from the reference and the culture sites were compared using mixed model analysis of variance (SAS statistical package, Release 8.0, GLM Procedure). Data were presented in box plots which indicated the full range of the data points as well the median values. Benthic macrofauna data were presented in terms of number of species, abundance and biomass (wet weight). An ANOVA testing for differences in the abundance and biomass of the phylum Annelida, between years and across locations was conducted.

3. Results

3.1. Sediment characteristics

3.1.1. Redox levels

Redox levels in the top 10 cm of sediment were not statistically different among sites ($p = 0.3$) or between replicates ($p = 0.26$) (Fig. 2). The mean redox level estimated over site, time and replicate was 226 ± 84 mV. In general, redox values were highest during the colder months, with negative values recorded only in August 2002 and August 2003.

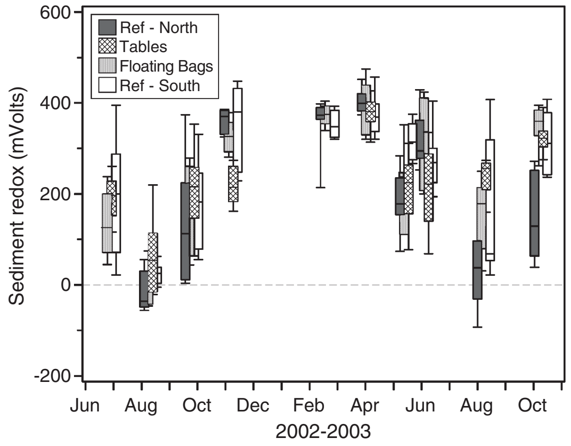

Fig. 2. Box plots of redox values (mV) in the top 10 cm of the sediment sampled at the two culture (hatched and grey bars) and two reference sites (solid and empty bars) from June 2002 to October 2003.

3.2. Sulfide levels

Estimates of sulfide levels in the upper 10 cm of sediment were not statistically different among sites ($p = 0.86$), but there was a significant replicate effect ($p < 0.01$) (Fig. 3). The mean sulfide level estimated over site, time and replicate was 733 ± 30 μM. No overall seasonal trend was apparent in the sulfide levels, but the greatest variability was observed at the Reference North and Table locations. There was no significant variation among sites in terms of the depth profile of sulfide (Site by Depth interaction: $p = 0.06$, Table 1). All sites generally exhibited higher sulfide levels in the surface layers (0–2 or 2–4 cm) than in the deeper layers (4–6, 6–8, 8–10 cm).

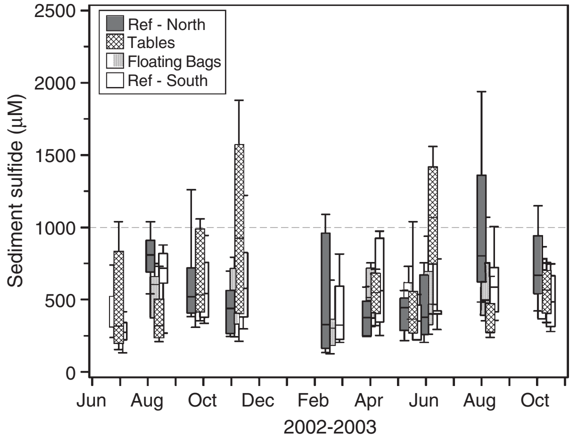

Fig. 3. Box plots of sulfide levels (μM) in the top 10 cm of the sediment sampled at the two culture (hatched and grey bars) and two reference sites (solid and empty bars) from June 2002 to October 2003.

Table 1
Least square mean sulfide (μM) levels at 2-cm intervals in the upper 10 cm of sediment from each site averaged over the duration of the study (June 2002–October 2003)

Sediment depth (cm)	Reference north sulfide (μM)	Floating bags sulfide (μM)	Oyster tables sulfide (μM)	Reference south sulfide (μM)
0–2	821 ± 69	648 ± 64	796 ± 73	610 ± 66
2–4	819 ± 71	671 ± 64	742 ± 70	678 ± 66
4–6	706 ± 69	521 ± 64	577 ± 70	581 ± 66
6–8	479 ± 69	410 ± 64	536 ± 70	398 ± 66
8–10	414 ± 69	323 ± 64	453 ± 73	332 ± 70

No significant differences were detected between the reference sites and the oyster culture sites.

3.3. Water content

Estimates of sediment water content in the upper 10 cm were statistically different among locations ($p < 0.01$) and across sampling times ($p < 0.01$), but the replicate effect was not significant ($p = 0.83$) (Fig. 4). Multi-mean comparison showed that the Table site had consistently lower percent water content (28.1%) than the other three locations which were statistically similar (37.8%) This in part reflects the coarser grain size or sandier characteristics of the sediment at the Table site (personal observation).

3.4. Organic content

Sediment organic content in the upper 10 cm also varied significantly among locations ($p < 0.01$) and across sampling times ($p < 0.01$), but not among

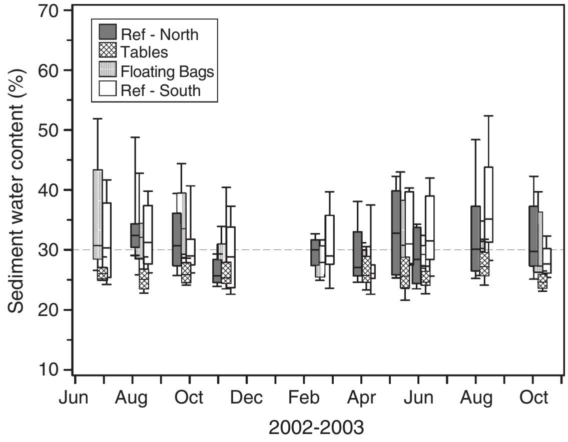

Fig. 4. Box plots of sediment water content (%) in the top 10 cm of the sediment sampled at the two culture (hatched and grey bars) and two reference sites (solid and empty bars) from June 2002 to October 2003.

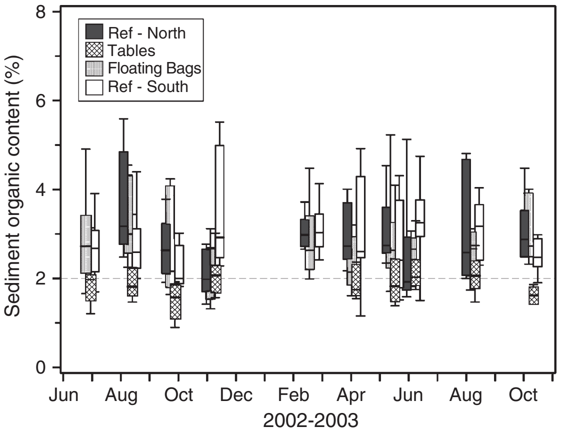

Fig. 5. Box plots of sediment organic content (%) in the top 10 cm of the sediment sampled at the two culture (hatched and grey bars) and two reference sites (solid and empty bars) from June 2002 to October 2003.

replicates ($p=0.35$) (Fig. 5). Multi-mean comparison showed that the Table site had a consistently lower percent organic content (2.2%) than the other three locations which were statistically similar (3.7%).

3.5. Sediment deposition

Rates of total sediment deposition in the sediment traps varied significantly among sampling sites ($p<0.01$) and over time ($p<0.01$) (Fig. 6). Neither the position of the trap (under or between the culture units) ($p=0.55$) nor the replication within a trap ($p=0.07$) were significant. Rates of organic sediment deposition followed the same pattern; significant differences were observed among sites ($p<0.01$) and over time ($p<0.01$), but there was no significant position ($p=0.8$) or replicate effect within a trap ($p=0.09$) (Fig. 7). Overall, the mean rate of sediment deposition at the Table location (81 g m^{-2} d^{-1}) was approximately double that recorded at the Reference sites (42.2 g m^{-2} d^{-1}). Similarly, the level of organic deposition at the Table location (16.5 g m^{-2} d^{-1}) was 75% higher than at the Reference sites (9.0 g m^{-2} d^{-1}). Despite these higher sedimentation rates, the mean organic content of the sediment deposited at the Table site (20.2%) was not significantly different from the Floating Bag (20.8%) or the Reference sites (21.8%). The highest sedimentation rate of 240 g m^{-2} d^{-1} was observed at all sites in September 2002 following a period of high wind and wave activity.

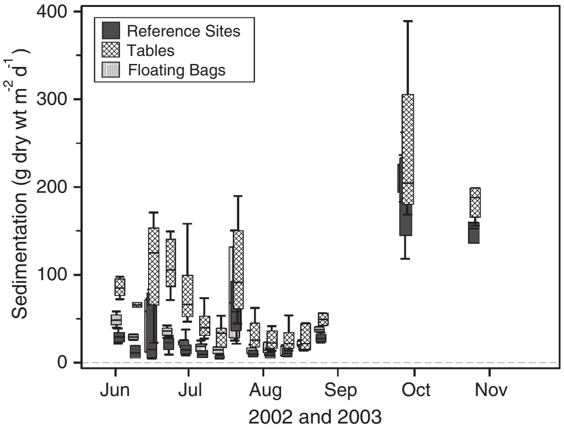

Fig. 6. Box plots of total sediment deposition rate (g dry wt m^{-2} d^{-1}) at the two culture sites (hatched and grey bars) vs. the mean of the reference sites (solid bar) averaged over the summers 2002 and 2003.

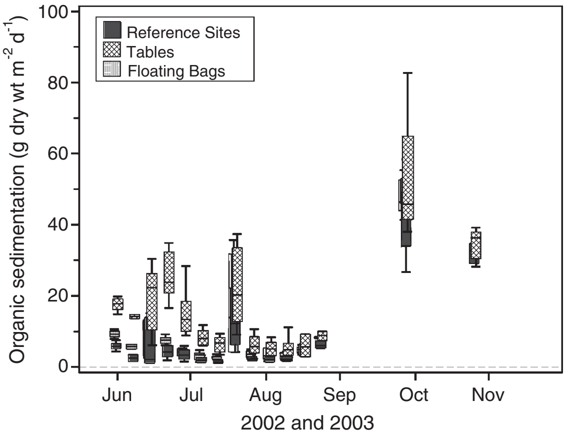

Fig. 7. Box plots of organic sediment deposition rates (g dry wt m^{-2} d^{-1}) at the two culture sites (hatched and grey bars) vs. the mean of the reference sites (solid bar) averaged over the summers 2002 and 2003.

3.6. Water column characteristics

No statistical difference was detected in chlorophyll levels ($p=0.5$) between the culture sites and the

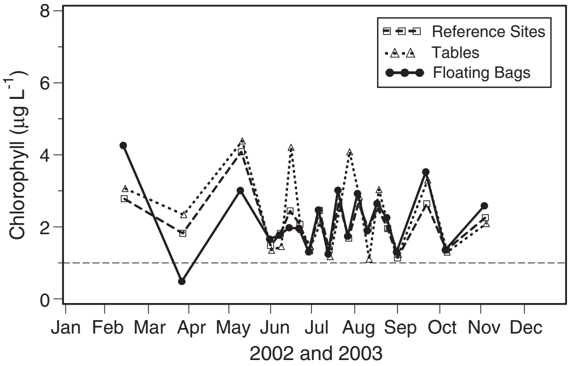

Fig. 8. Chlorophyll levels (µg L^{-1}) at the two culture sites vs. the mean of the reference sites averaged over 2002 and 2003.

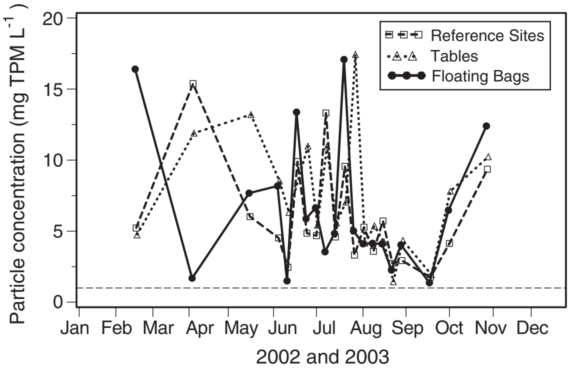

Fig. 9. Concentration of total particulate matter (mg TPM L^{-1}) at the two experimental sites and the mean of the reference sites averaged over the two years.

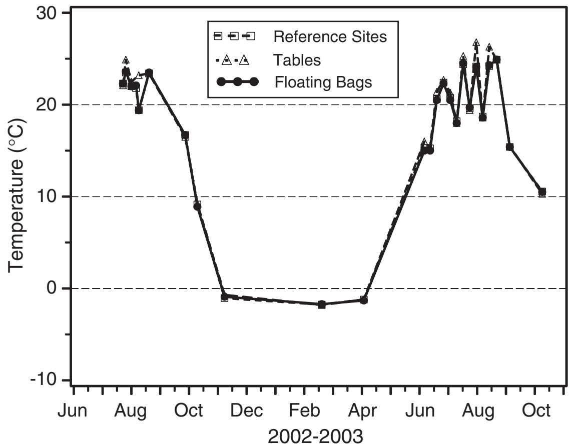

Fig. 11. Temperature (°C) recorded at the two experimental sites and the mean of the reference sites from June 2002 to October 2003.

reference sites (Fig. 8). The overall mean chlorophyll was 2.0±0.5 μg L^{-1} with values ranging from 1.0 to 4.0 μg L^{-1}. No statistical difference was detected in water column TPM ($p=0.5$) or POM levels ($p=0.5$) between the culture sites and the reference sites (Figs. 9 and 10). Variations among sites during the winter sampling period may be an artefact due to disturbance of benthic sediments while cutting through the ice.

Profiles of temperature and salinity (Figs. 11 and 12) were similar for the culture and reference sites suggesting that this region of the bay is well-mixed. Temperature levels above 20 °C were observed for 4 months of the year (June to September), whereas temperatures below 4 °C were recorded for 6 months (November to April). The salinity increased substantially during the winter months, probably due to the 1-m layer of ice coverage which inhibits freshwater land drainage as well as restricting exchange within the bay.

3.7. Benthic community composition

A total of 25 taxa were recorded including 5 polychaetes, 4 crustaceans, 10 bivalves, 5 gastropods and 1 fish species. These taxa were assigned to five functional trophic groups including deposit feeders, herbivores, omnivores, predators and suspension feeders. The number of species, abundance and biomass of organisms in each trophic group at each site are indicated in Table 2. Note that these data represent one set of replicated sediment samples obtained at each of the four sites in September 2002 and 2003. Also note that the biomass estimates can be easily skewed by the presence of a single large organism; for example, in several instances the presence of one rare bivalve species accounted for 95% of the wet weight of the suspension

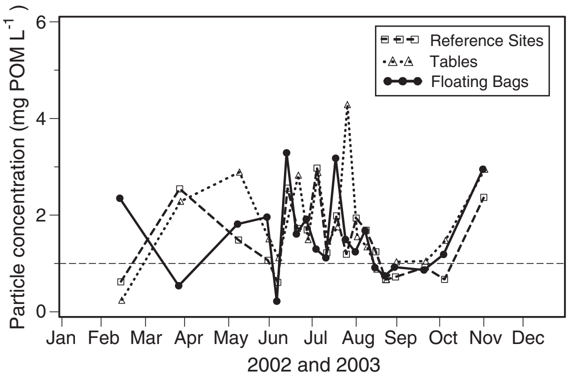

Fig. 10. Concentration of particulate organic matter (mg POM L^{-1}) at the two experimental sites and the mean of the reference sites averaged over the two years.

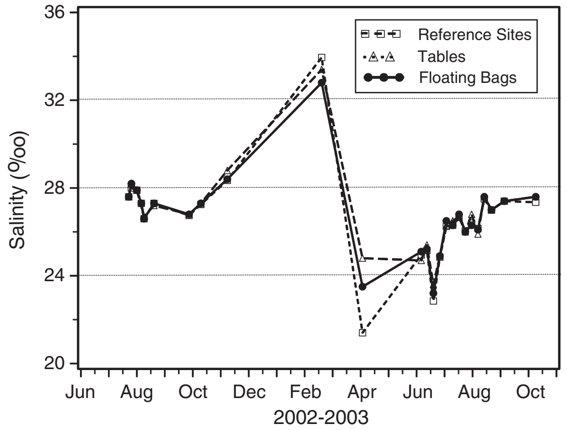

Fig. 12. Salinity (‰) recorded at the two experimental sites and the mean of the reference sites from June 2002 to October 2003.

feeders. Even if the contribution of the shell were to be removed, these rare individuals would dominate the biomass index. The species which accounted for a large proportion of the abundance at each site was the small herbivorous gastropod *Bittium alternatum* (1–2 mm). Two other prominent species observed at all sites were the omnivorous polychaete *Nereis diversicolor* and the deposit-feeding thread worm *Heteromastus filiformis*. The most abundant suspension feeder was the small clam *Gemma gemma* (2–3 mm).

An analysis of variance using the Annelida phylum as a reference benthic group showed a significant difference in biomass between years but not among sites (Table 3). Several oyster growers in eastern Canada observed oyster mortalities during the winter 2002–2003, and a decline in the abundance of the benthic organisms may be related to severe environmental conditions. The biomass of the Annelida phylum decreased by 41% between September 2002 and September 2003, with the highest decline recorded at the Table site in the intertidal zone (55%). During the same period, annelid abundance increased by 56%, with the lowest increase recorded at the Table site (12%) and the highest increase at the Reference South Site (100%). This could reflect a die-off of larger specimens during the winter followed by juvenile recruitment in 2003. Given the extended winter and severe ice conditions, substantially more data from the beginning and end of each growing season would be required to properly describe the trends in the benthic community.

Table 3
ANOVA testing for differences in the abundance and biomass of the phylum Annelida, between years and across locations

Source	Abundance			Biomass	
	DF	MS	Pr<*F*	MS	Pr<*F*
Year (Y)	1	115,801	0.06	1059	0.01
Location (L)	6	17,103	0.73	102	0.62
Type (Y L)	8	52,090	0.15	932	<0.01
Error	31	28,887		137	

Table 2
Description of the five trophic groups in the benthic community at each site in September 2002 and 2003, in terms of number of species, abundance and biomass (wet weight)

Location	Trophic group	2002			2003		
		Species number	Abundance (ind m^{-2})	Biomass (g m^{-2})	Species number	Abundance (ind m^{-2})	Biomass (g m^{-2})
Reference north	Dep	4	262	29.9	4	275	10.2
	Herb	1	2212	7.2	1	1387	4.9
	Omni	4	400	34.7	4	675	35.3
	Pred	2	50	0.5	0		
	Susp	2	50	4.8	2	63	7.5
	Total	13	2974	77.1	11	2337	50.4
Floating bags	Dep	3	275	14.9	4	600	16.7
	Herb	1	1350	4.9	1	338	1.2
	Omni	2	338	67.1	4	462	37.6
	Pred	2	850	3.2	0		
	Susp	5	62	12.3	4	87	203.1
	Total	13	2875	102.4	13	1487	258.6
Oyster tables	Dep	3	488	16.5	3	675	22.9
	Herb	1	4175	16.5	1	762	2.7
	Omni	6	625	150	3	275	5.4
	Pred	2	175	1.7	1	50	82
	Susp	3	425	63	2	87.5	1.5
	Total	15	5888	248.0	10	1850	115.0
Reference south	Dep	3	125	26.8	3	388	9.7
	Herb	1	512	1.9	1	1700	5.8
	Omni	1	212	21.7	3	500	25.5
	Pred	3	75	65.6	1	75	0.5
	Susp	4	337	132	4	88	175.3
	Total	12	1261	248.0	14	2751	216.8

Dep=Deposit feeders, Herb=Herbivores, Omni=Omnivores, Pred=Predators, Susp=Suspension feeders.

4. Discussion

In Canada, "environmental impact" in the context of the marine ecosystem is defined as the alteration or destruction of fish habitat leading to significant changes in the overall productivity of the affected ecosystem (DFO, 1986). Significant environmental impacts associated with shellfish culture have been documented in Europe (Dahlback and Gunnarsson, 1981), South Africa (Stenton-Dozey et al., 1999) and New Zealand (Kaspar et al., 1985), but these impacts must be viewed in the context of the intensity of the culture activities and the physical conditions at the site. For example, mussel raft culture in South Africa produces up to 27 tons of product annually per raft (14 × 11 m) or the equivalent of 175 kg m^{-2} (Stenton-Dozey et al., 1999). In Sweden, 100 tons of mussels were reportedly produced in an 18-month period in a lease area of 2800 m^2, or the annual equivalent of 24 kg m^{-2} (Dahlback and Gunnarsson, 1981). In comparison, the annual production for a mussel culture lease in Prince Edward Island (Canada) is estimated at 3 kg m^{-2}, and an oyster culture lease in northern New Brunswick has an annual production of approximately 0.4 kg m^{-2}. This relatively low annual production rate is a function of the low oyster stocking densities as well as the lack of growth during the 6-month winter period.

The St-Simon culture site is a shallow open bay with excellent water exchange, an extensive eelgrass bed, a healthy benthic community, oxic sediments, low sediment organic matter and mean sulfide levels less than 1000 μM. Bottom sediments are frequently subjected to re-suspension by wind events, particularly during the spring and fall, and physical erosion by ice contact during the winter months. Given the dynamic environment and the low production intensity, it is not surprising that this study indicated no measurable impact of oyster culture on the benthic sediment chemistry. In a similar study in Tasmania, Crawford et al. (2003) concluded that oyster culture had little impact on the benthic environment and that extensive monitoring was unnecessary. In Prince Edward Island where the most intensive form of shellfish aquaculture is being practiced in eastern Canada, a benthic sediment survey of 20 estuaries showed no specific impact of mussel culture activities; anoxic sediments and high levels of organic matter were typical of both cultured and culture-free sites (Shaw, 1998). Likewise in Nova Scotia, Grant et al. (1995) documented no negative impact of mussel culture on the benthic community, with the exception of a partial shift toward anaerobiosis under the mussel lines.

In the present study, one parameter which did differ significantly among sites was the level of sedimentation. The mean estimate for the Floating Bag site (50 g m^{-2} d^{-1}) was 25% higher than the Reference sites (42 g m^{-2} d^{-1}), while the estimate for the Table site (81 g m^{-2} d^{-1}) was 93% higher. Note that the raft culture of mussels in Spain is associated with sediment deposition values on the order of 515 g m^{-2} d^{-1} (Perez-Camacho et al., 1991). Increased sedimentation levels are typically documented under mussel culture operations, but this additional input does not necessarily result in higher sulfide levels and/or community degradation. Tita et al. (2003) noted that sedimentation rates were 50% higher at a mussel culture site than at a reference site (Magdalen Islands, Canada), but no changes were detected in the benthic macrofauna community. Grant et al. (1995) also observed 50% higher sedimentation rates under the mussel lines relative to the reference site, but this was associated with a localized increase in macrofauna biomass.

Although the oysters may have contributed to the relatively high deposition rate at the Table site (81 g m^{-2} d^{-1}), the organic content of the sediment collected in the traps (20%) was not significantly higher than at the other sites. It may be argued that the natural sedimentation rate was likely enhanced by the hydrodynamic interference of the oyster tables (Ottman and Sornin, 1985; Nugues et al., 1996). In bays with very high particle loads such as the Marennes-Oléron Basin in France (20 to 200 mg L^{-1}; Héral, 1991), the presence of oyster tables led to substantial accumulation of sediments which had negative impacts on sediment biochemistry. Crawford et al. (2003) noted that the variation in sedimentation levels among three different oyster culture sites was greater than that observed between the culture and reference locations within a site, thus reflecting the importance of localized factors such as current flow patterns.

In the present study, the organic sedimentation rate at the Table site was 16.5 g m^{-2} d^{-1} compared to 9.0 g m^{-2} d^{-1} at the Reference sites. Despite this higher organic sedimentation level, the organic content in the sediment at the Table site (2%) was consistently lower than at the other sites (3–5%). This suggests that the organic matter in the biodeposits was not being incorporated into the sediments, but was either washed away and/or rapidly processed by the benthic community. Martin et al. (1991) showed that in cases where an accumulation of biodeposits was detected in sandy sediments, this impact was undetectable 2 months after the oyster tables had been removed. The natural concentration of organic matter in the sediment at a particular site and the potential for the accumulation of additional organic matter are likely important factors in assessing the possible impact of shellfish culture. For example, compared to the organic content of the sediment at the New Brunswick site (2–5%), the sediment of both culture-free and cultured sites in Prince Edward Island estuaries have a relatively high organic content (9–11%) (Shaw, 1998). Note that Chivilev and Ivanov (1997) observed a degradation of the benthic community when the concentration of organic matter exceeded 9–10%.

The higher level of organic sedimentation at the Table site may account for the greater abundance of deposit feeders, particularly in 2002. In general, the number of species and macrofaunal abundance was similar at the culture and the reference sites, and there was no evidence of species associated with highly disturbed areas such as *Capitella capitata*. The range of estimates for macrofaunal abundance (1261–5888 ind m^{-2}) was similar to estimates (61–6059 ind m^{-2}) reported in a survey of four wild oyster beds on the eastern New Brunswick coast (Milewski and Chapman, 2002). Thorne (1998) found that intertidal oyster culture areas in Tasmania had a higher species number, diversity, and abundance than reference areas. Similarly, Dealteris et al. (2004) observed that oyster cages placed on the seabed supported a significantly higher abundance of organisms per m^2 than either reference areas with aquatic vegetation or non-vegetated seabed. Nugues et al. (1996) detected small but significant differences in the macrofaunal community located directly below oyster tables compared to that found in adjacent uncultivated areas. In contrast, significant decreases in macrofaunal abundance have been documented in areas of extensive intertidal oyster culture in France (Castel et al., 1989).

It should be noted that the development of epifaunal fouling communities on the culture gear and the shellfish stock may compound the impacts associated with the cultured species. In the Thau lagoon in southern France, the presence of fouling organisms may contribute to the incidence of oxygen depletion events (Mazouni et al., 1998, 2001). In South Africa, the presence of the ascidian *Ciona intestinalis* and other fouling organisms on the mussel rafts accounted for an additional 78 tons of annual wet weight production above the 27 tons of mussels (Stenton-Dozey et al., 1999). The faecal matter associated with this substantial fouling biomass likely contributed significantly to the high biodeposition rates which were implicated in the degradation of the benthic community.

In conclusion, the study suggested that the level of oyster culture (8 kg m^{-2}) currently practiced in this dynamic shallow water environment is not sufficient to negatively impact either the sediment biochemistry or the associated benthic community. With increasing culture activity, however, continued environmental monitoring is important to ensure that the overall health of the ecosystem is maintained. Redox and sulfide measurements may not be the most appropriate indices for assessing the impact of oyster culture, and other indices such as benthic community structure and/or sediment organic content may prove to be more useful.

Acknowledgements

We would like to thank the New Brunswick Shellfish Aquaculture Environmental Coordinating Committee for supporting this project. Funding for the study was provided by the Aquaculture Collaborative Research and Development Program of the Department of Fisheries and Oceans, the Professional Shellfish Growers of New Brunswick, and Mallet Research Services Ltd. The New Brunswick Department of Agriculture, Fisheries and Aquaculture provided laboratory space and equipment, and the oyster company, L'Étang Ruisseau Bar Ltée, provided technical support. Special thanks are extended to Lisa Doucette and Dr. Barry Hargrave (Department of Fisheries and Oceans) for providing instruction and guidance on the analytical procedures for assessment of sediment redox and sulfide levels. We also thank summer students Julie Mallet, Olivier Dugas, and Sabrina Duguay who participated in the collection and processing of the samples.

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Aquaculture 287 (2009) 315–323

Contents lists available at ScienceDirect

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journal homepage: www.elsevier.com/locate/aqua-online

The effect of floating bag management strategies on biofouling, oyster growth and biodeposition levels

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ARTICLE INFO

Article history:
Received 28 June 2007
Received in revised form 13 October 2008
Accepted 14 October 2008

Keywords:
Biodeposition
Biofouling
Crassostrea virginica
Environmental impact
Oyster culture

ABSTRACT

Commercially-viable aquaculture of the Eastern oyster (*Crassostrea virginica*) in New Brunswick, Canada, depends on optimizing production strategies during the short growing season while minimizing environmental impacts. Fouling levels on the floating bags used for oyster culture are typically controlled by periodic turning in order to expose the accumulated biomass to air drying. The impact of different bag turning frequencies on fouling levels, oyster production, biodeposition rates and the settlement of fouling species was assessed at two sites in northern New Brunswick. Over the 4 month study period (June–October 2006) the fouling biomass increased exponentially to a maximum of 300 g dry wt. for bags which were never turned. Maintenance activity aimed at reducing fouling levels did not, however, enhance oyster performance; growth and survival rates were similar in bags turned bi-weekly, monthly, once in mid-August or never. Biodeposition rates under the bags were also similar among handling treatments and did not significantly exceed reference levels even immediately following bag turning events. With regard to fouling control, bag turning was effective in eliminating the settlement of barnacles and mussels on the bags, but not on the oysters. In general, given the current fouling community, turning floating bags once in mid-August following the barnacle settlement in late July would likely be sufficient to control most species. Ongoing mussel settlement during September may, however, necessitate one additional handling in early October prior to sinking of the bags for the winter.

1. Introduction

Aquaculture of the Eastern oyster (*Crassostrea virginica*) in Atlantic Canada has grown significantly in the past decade, largely as a result of the introduction of floating bag culture. This technique has increased oyster production rates to the point where commercial size can be attained within 4 years, as opposed to 5 to 8 years when grown on the bottom (Anonymous, 2003). In the Gulf of St. Lawrence region, oyster production has increased from approximately 2500 t in 1994 to 5000 t in 2005 (Anonymous, 2005), and a considerable expansion is expected in the short-term. Although growing oysters in floating bags has been found to have negligible impacts on the benthic environment (Mallet et al., 2006), handling strategies for preventing the accumulation of fouling organisms on these structures may have implications for oyster productivity and biodeposition levels.

Optimization of culture equipment and husbandry methods are two critical elements for improving productivity in shellfish farming (Robert et al., 1993; Handley 2002; Louro et al., 2007). In the case of oyster culture, the location of the farm (e.g. intertidal or subtidal) and the winter conditions (e.g. presence or absence of ice) pose constraints as to the choice of gear. In northern New Brunswick (Canada), floating oyster bags are currently the preferred culture method; these flat UV-resistant polymer bags are equipped with floats which keep them semi-submerged at the surface of the water. During the winter months when there is extensive ice coverage (November to April), producers can conveniently remove the floats, allow the bags to sink to the bottom and re-float them the following spring. In the summer months, fouling levels can be controlled by turning the bags to expose the previously-submerged portion to the air and sunlight; growers can thus effectively eliminate fouling organisms on the culture gear while the oysters remain submerged.

Minimizing the development of fouling assemblages on caged structures has been generally shown to have a positive impact on the growth of cultured shellfish (Michael and Chew, 1976; Cropp and Hortle, 1992; Claereboudt et al., 1994a; Lodeiros and Himmelman, 1996; Taylor et al., 1997). It has been argued that the fouling of culture equipment reduces water flow thereby restricting particle flux or rates of food supply and waste removal (Wallace and Reisnes, 1985; Mouland and Parsons, 1999; Claereboudt et al., 1994b). Given the relatively short growing season in northern New Brunswick it is essential to ensure that the oysters have unimpeded access to the available food. Depending on the composition of the fouling community, there may also be significant competition for food

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doi:10.1016/j.aquaculture.2008.10.023

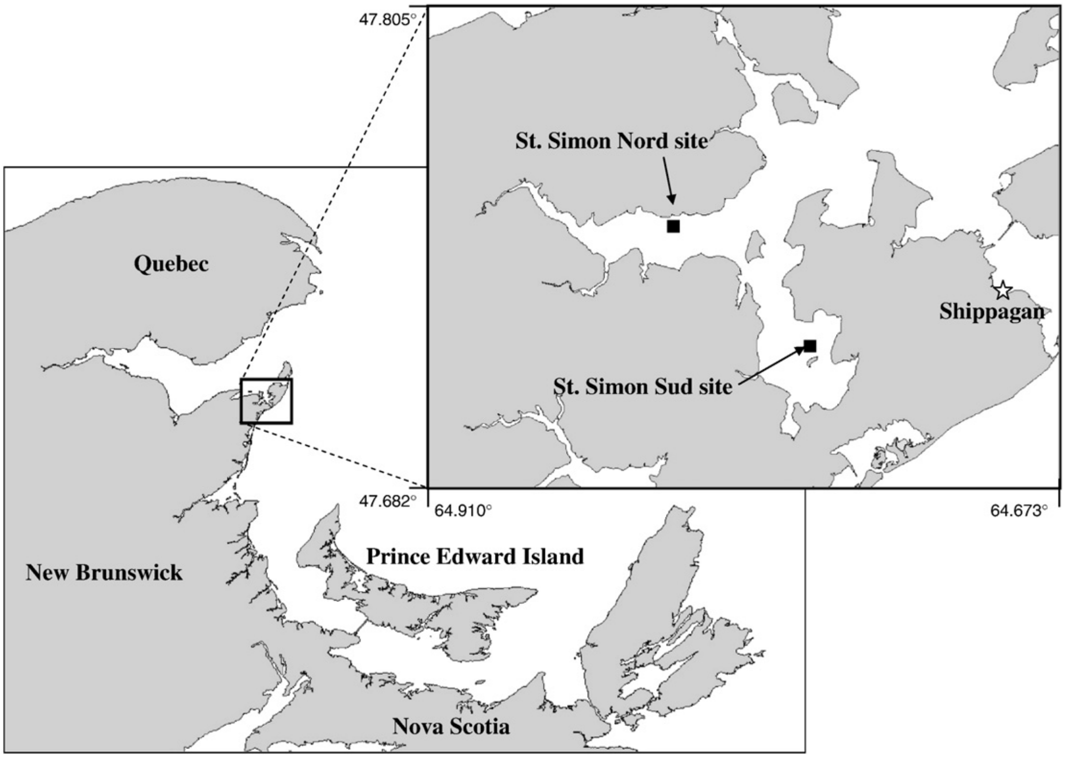

Fig. 1. Location of the oyster culture leases in Baie St-Simon Nord and Baie St-Simon Sud on the northeast coast of New Brunswick, Canada.

resources from other filter-feeders. Although studies have demonstrated the positive impacts of reducing net or tray fouling on the performance of sea scallops (Claereboudt et al., 1994a,b) and pearl oysters (Southgate and Beer, 1997; Taylor et al., 1997; Pit and Southgate, 2003), there are no studies reporting on the impact of fouling control strategies on the growth of Eastern oysters in floating bag culture. Another concern with regard to the accumulation of fouling organisms on the grow-out structure is the potential environmental impact associated with increased levels of biodeposition which may be linked to benthic enrichment. If not managed appropriately, this assemblage may contribute substantially to the level of biodeposition, both in terms of faecal matter as well as fall-off during maintenance and harvesting (Stenton-Dozey et al., 2001). This potentially important issue has received little attention in previous studies on the environmental impacts of shellfish culture. Assessing biodeposition levels is therefore an important aspect of determining the most appropriate

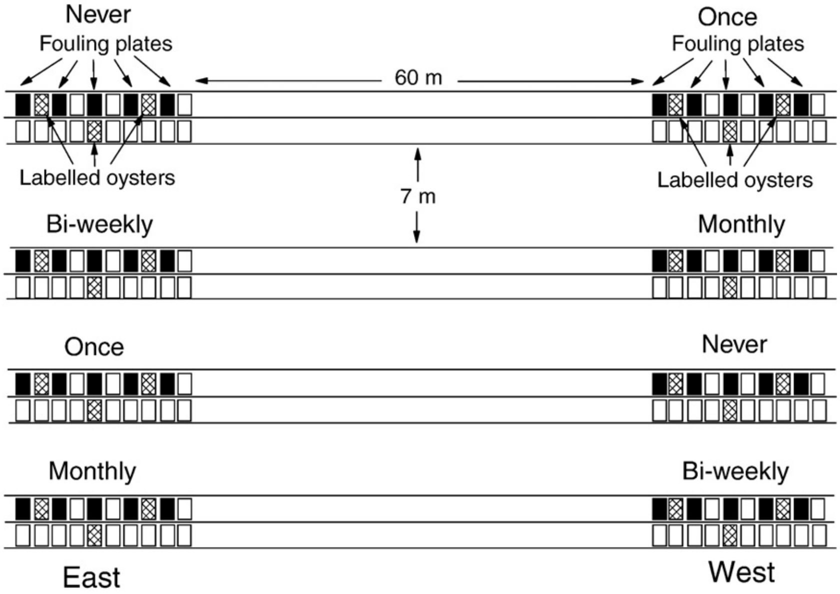

Fig. 2. A description of the experimental setup at each of the two sites in the study. Each of the four treatments was established on the east and the west end of four adjacent longlines oriented in an east–west direction. These double 100 m longlines normally hold 100 oyster bags, but in this case only 20 experimental bags were attached at either end. Note that bags at the same position in each treatment were used for either monitoring oyster performance (3-hatched) or deploying the fouling plates (5-black). An additional series of cumulative fouling plates was attached to the 15 unused bags in each of the “Never” treatments.

Table 1
ANOVA testing the effect of site (*S*), handling frequency (*H*) and their interaction on the accumulation of fouling biomass on the oyster bags

Source	DF	Fouling biomass	
		MS	Pr>*F*
Site (S)	1	2691	<0.01
Handling (H)	3	12614	<0.01
S×H	3	75.3	0.36
Rep (S H)	8	62.5	<0.01
Error	64	21.9	

Data were rank-transformed because variances were heterogeneous (Bartlett $p<0.01$).

handling regime for controlling fouling levels on floating oyster bags (Bastien-Daigle et al., 2007).

Although minimizing biofouling may improve product yield and reduce the risk of environmental impacts, the removal of this biomass on a commercial scale is a costly activity (Young-Lai and Aiken, 1986; Gribben et al., 2006). Decisions with regard to the frequency and timing of handling and/or the degree of effort invested in controlling fouling levels may be critical in determining “operating costs and profitability” (Pit and Southgate, 2003). Hence it is important to compare the impact of various handling strategies in order to determine the best maintenance regime for a particular site given the intensity of fouling and the composition of the fouling assemblage. An important aspect of the present study was to assess the impact of different handling strategies on the recruitment success of two major competitors, specifically barnacles (*Balanus improvisus*) and mussels (*Mytilus edulis/trossulus*).

2. Materials and methods

2.1. Location of study and experimental setup

The field study was conducted at two sites in Baie St-Simon, a large inlet situated near Shippagan in northern New Brunswick, Canada (Fig. 1). One site was located in the northern arm of the bay (St-Simon Nord), and the second site was located in the southern arm of the bay (St-Simon Sud). Four 100 m longlines were set up at each site and twenty 9 mm mesh Intermas floating bags containing 200 oysters each were deployed at the far east and west end of each line for a total of four groups of 20 bags on either side or 160 bags/site (Fig. 2). The four replicated groups on each side were then assigned to four handling frequencies or treatments: (a) turned “Biweekly” or every other week, (b) turned “Monthly” or every 4 weeks, (c) turned “Once” during mid-August, and (d) “Never” turned. The various treatments were assigned such that the replicated handling frequencies never occurred on the same line. The start date of the study was June 14, 2006 and the end date was October 18, 2006.

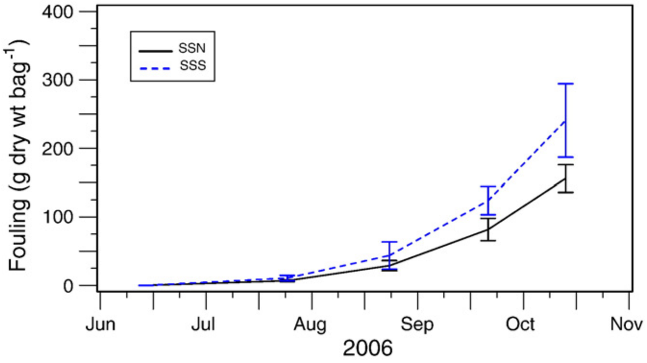

Fig. 3. Profile of the accumulation of fouling biomass (mean ± 2 SE) on the “Never”-turned bags at the two sites.

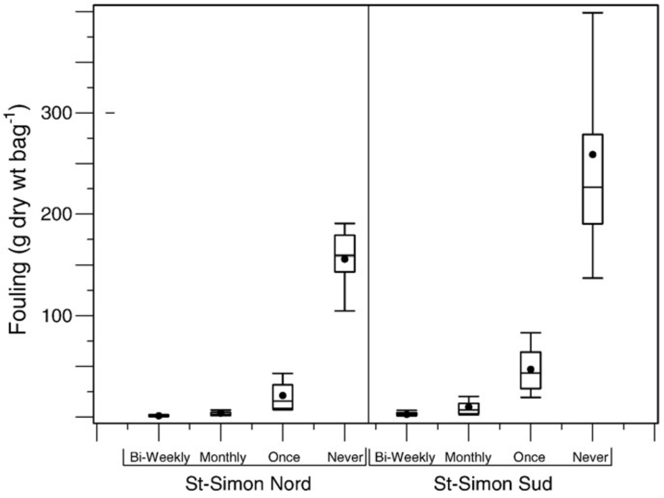

Fig. 4. Biomass (dry weight) of fouling organisms on the oyster bags at the two sites for each of the four handling frequencies. The solid circle represents the mean value, the crossbar inside each box indicates the median, and the bottom and top whiskers represent the 10th and 90th percentile, respectively.

2.2. Biofouling: biomass and species monitoring

Settlement plates (13 × 15 cm) cut from used 9 mm mesh oyster bags were cleaned, dried at 60 °C for 48 h and weighed. Plates were then attached to the submerged side of 5 of the 20 oyster bags in each of the replicated “Biweekly”, “Monthly”, “Once” and “Never” handling treatments to assess the accumulation of biomass over each sampling interval (Fig. 2). Immediately prior to each bag turning event, new biomass plates were attached to the upper side of the same 5 bags in the appropriate handling replicate. All 20 bags were then turned and the 5 submerged plates from the previous interval were removed. The plates were placed in individual containers filled with seawater which were then transferred to a cold room (4 °C) and processed within 12 h of retrieval. Each plate was weighed to determine wet weight, and then dried at 60 °C (48 h) and re-weighed to determine the dry weight of organisms. Control plates (13 × 15 cm) of known weight were also dipped in seawater at the time of sampling and processed similarly to account for the weight of dried salts.

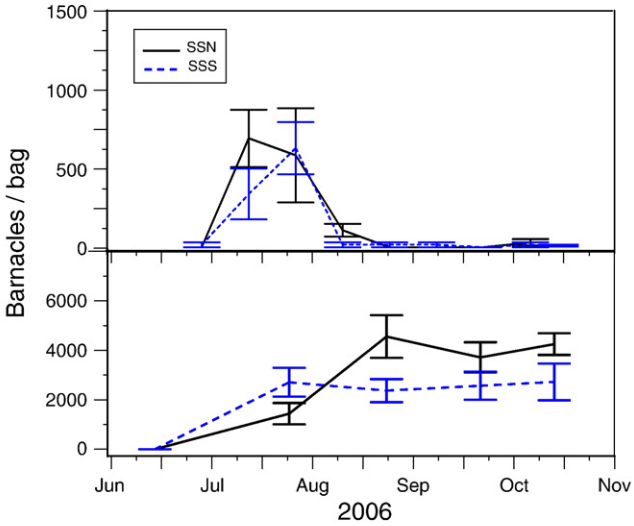

Fig. 5. “Bi-weekly” (top) and cumulative (bottom) settlement of barnacles (mean ± 2 SE) on the oyster bags at the two sites. Each point in the “Biweekly” profile represents the settlement observed within a 2 week period. The cumulative settlement data originate from plates removed every month from the “Never” handled bags.

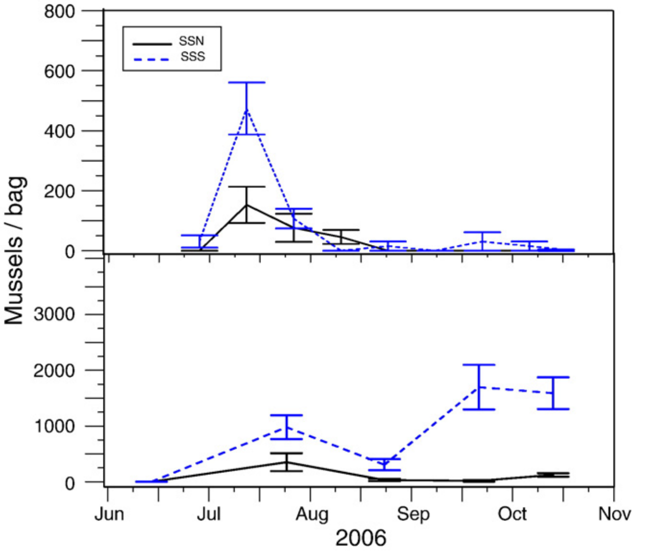

Fig. 6. “Bi-weekly” (top) and cumulative (bottom) settlement of mussels (mean ± 2 SE) on the oyster bags at the two sites. Each point in the “Biweekly” profile represents the settlement observed within a 2 week period. The cumulative settlement data originate from plates removed every month from the “Never” handled bags.

At the start of the study additional pre-weighed biomass plates were attached to the submerged side of each of the 15 unused bags in the “Never” handled treatment (Fig. 2). Five of these plates were removed and processed in mid-July, mid-August, and mid-September to document the accumulation of fouling biomass over time; note that these “cumulative” plates were removed gently without turning the bags. Estimates of biomass/plate (13×15 cm = 195 cm^2) were converted to biomass/bag assuming that only the lower half of the bag was submerged (surface area of the submerged section = 0.529 m^2).

In each “Biweekly” handling treatment, 5 species monitoring plates (10×10 cm) were attached along with the biomass plates (1/bag) to provide a standardized settlement surface for barnacles and mussels. These plates were removed and replaced every 2 weeks following the same sampling procedure as the “Biweekly” biomass plates. In addition, a series of cumulative species settlement plates were attached to each of the 15 extra bags in the “Never” treatment (1 per bag) along with the cumulative biomass plates; 5 of these settlement plates were removed every month to document the development of the barnacle/mussel population over time. The plates were stored in a cold room (4 °C) until they could be assessed for barnacle and mussel abundance within 48 h of sampling. Six fields on each settlement plate (2.4×2.4 cm each) were enumerated under a dissecting microscope (16x). Mean estimates of the number of barnacles or mussels per field (5.76 cm^2) were converted to number of individuals per bag (0.529 m^2).

The settlement of barnacles and mussels on the oysters inside the bags was also evaluated at the end of the study. Ten oysters were randomly sampled from 6 of the 20 bags in each handling replicate and the number of barnacles and mussels/individual oyster was determined.

Table 2
ANOVA testing for the effect of site (*S*) and handling frequency (*H*) and their interaction on the abundance of barnacles and mussels on the bags

Source	DF	Barnacles		Mussels	
		MS	Pr > *F*	MS	Pr > *F*
Site (S)	1	909.5	0.03	839.8	<0.01
Handling (H)	3	14232.7	<0.01	526.1	<0.01
S × H	3	325.1	0.16	371.2	<0.01
Rep (S H)	8	152.1	0.82	6.2	0.99
Bag (R S H)	163	281.9	<0.01	27.8	<0.01
Error	405	25.1		11.4	

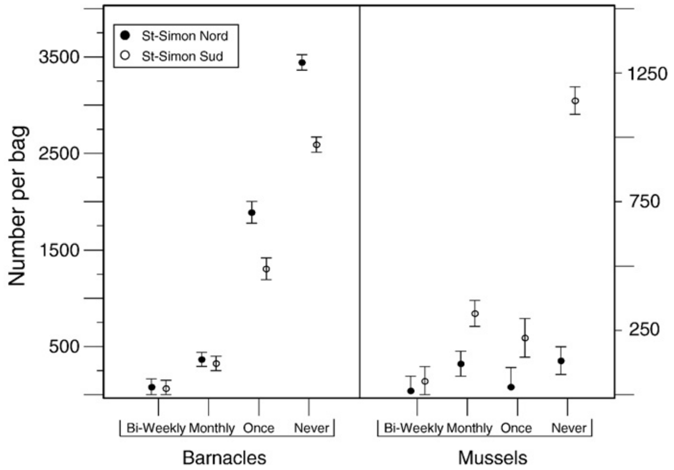

Fig. 7. Number (mean ± 1 SE) of barnacles and mussels/bag at each site for each handling frequency. Note the right-hand axis for the number of mussels.

2.3. Oyster performance

Near market-size 3 year-old oysters (55–60 mm) from a common lot were distributed among the various treatments (2 sites × 160 bags/site × 200 oysters/bag or 64,000 oysters) in early June 2006 during the initial setup of the sites. To evaluate growth performance 48 groups of 10 oysters were individually labelled and measured (shell height, width, length and whole weight) to establish their pre-handling status. On June 14, groups of 10 labelled oysters were placed in 3 of the 20 bags in each replicated handling treatment at each site (2 sites × 4 treatments/site × 2 replicates/treatment × 3 bags/replicate × 10 oysters/bag or 480 labelled oysters) (Fig. 2). The oyster bags were manipulated as per the handling schedule and the labelled individuals were reassessed at the conclusion of the study on October 18 2006.

2.4. Biodeposition

The impact of biofouling relative to background deposition rates was assessed by deploying sediment traps for 24–48 h prior to handling the “Monthly”, “Once” and “Never” treatments. These traps (two 6.6 cm diameter tubes/trap) were retrieved, emptied and redeployed immediately before the handling event and then retrieved 48 h post-handling. During each monitoring period (pre- and post-handling) two reference traps were deployed 50 m away from the experimental lines to document variations in the natural biodeposition or sedimentation rate (see Mallet et al., 2006). Sediment traps were also deployed under the “Never” treatment bags at monthly intervals to document the relationship between biodeposition and increasing fouling levels. Material collected in the traps was evaluated for dry weight (total and organic) as described in Mallet et al. (2006). Estimates of biodeposition were converted from sediment dry weight (g)/replicate tube (area: 34.2 cm^2)/h to g m^{-2} d^{-1}.

Table 3
ANOVA testing the effect of site (*S*) and handling frequency (*H*) and their interaction on the abundance of mussels and barnacles on the oysters

Source	DF	Barnacles		Mussels	
		MS	Pr > *F*	MS	Pr > *F*
Site (S)	1	25.3	0.12	0.76	0.05
Handling (H)	3	14.9	0.22	0.56	0.04
S × H	3	18.5	0.16	0.25	0.25
Rep (S H)	8	8.3	0.02	0.14	0.52
Oyster (R S H)	79	3.3	<0.01	0.14	0.84
Error	826	0.5		0.24	

Abundance data were log-transformed because variances were heterogeneous (Bartlett $p < 0.01$).

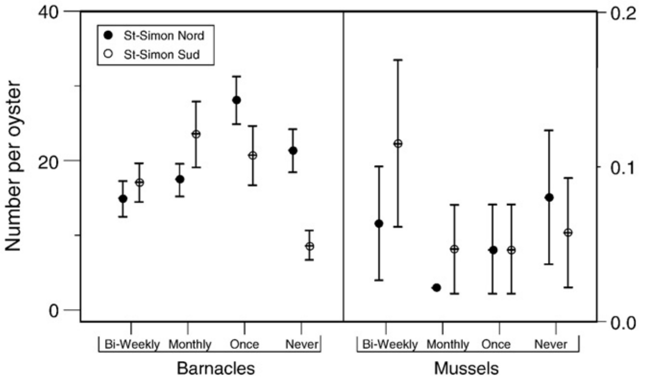

Fig. 8. Number (mean ± 1 SE) of barnacles and mussels/oyster at each site for each handling frequency. Note the right-hand axis for the number of mussels.

2.5. Environmental monitoring

Vemco temperature probes, set to record hourly, were deployed at each site for the duration of the study. At bi-weekly intervals approximately 1 L of seawater was collected from 10 cm below the surface for measurements of chlorophyll. Details of the chlorophyll analysis are provided in Mallet et al. (2006).

2.6. Statistical analysis

The following mixed model analysis of variance (SAS ver. 9.1, procedure GLM) was used to test the main fixed effects (Site (i = 1,2), Handling frequency (j = 1 to 4)) and their interactions on oyster growth in terms of shell height or whole weight. Rep (k = 1,2), a random effect, represents the replicated treatment located on the east and west side of each site. Bag (l = 1 to 3), a random effect, represents the three replicated bags, each containing 10 individually labelled oysters, found in each replicated treatment. Bartlett's test was used to check for homogeneity of variance; the data were transformed if the assumption of homoscedasticity was not met (see Table captions).

$$\begin{aligned}\text{Growth (Height or Weight)}_{ijklm} = {} & \text{u} + \text{Site}_i + \text{Handling}_j + \text{Site} \\ & \times\text{Handling}_{ij} + \text{Rep}_k(\text{Site}_i\ \text{Handling}_j) \\ & + \text{Bag}_l(\text{Site}_i\ \text{Handling}_j\ \text{Rep}_k) \\ & + \text{Error}_{ijklm}\end{aligned}$$

The following mixed model analysis of variance was used to test both for differences in fouling biomass (Plate) and biodeposition levels (Trap) caused by the main fixed effects (Site, Handling frequency) and their interactions. The replication within site and treatment (Rep) and the Plate or Trap were declared as random effects.

$$\begin{aligned}\text{Biodeposition or Fouling biomass}_{ijklm} = {} & \text{u} + \text{Site}_i + \text{Handling}_j \\ & + \text{Site}\times\text{Handling}_{ij} \\ & + \text{Rep}_k\,(\text{Site}_i\ \text{Handling}_j) \\ & + \text{Plate}_l\,(\text{Site}_i\ \text{Handling}_j\ \text{Rep}_k) \\ & + \text{Error}_{ijklm}\end{aligned}$$

The following mixed model analysis of variance was used to test for differences in biodeposition levels associated with the presence or absence of culture activity (i.e. Culture vs Reference sites). In this case Site, Handling frequency, Time and Trap were declared random effects.

$$\begin{aligned}\text{Biodeposition}_{ijklm} = {} & \text{u} + \text{Site}_i + \text{Culture}_j + \text{Handling}_k \\ & + \text{Time}_l\,(\text{Culture}_j\ \text{Handling}_k) \\ & + \text{Trap}_m + \text{Error}_{ijklm}\end{aligned}$$

3. Results

3.1. Biofouling: biomass accumulation

Statistical analyses were carried out on the mean biomass values obtained for the five bags within each replicated treatment at each site. The accumulation of fouling biomass (dry weight) on the submerged surface of the oyster bags was significantly affected by Site, Handling frequency and their interaction (Table 1). In general, higher levels of fouling were observed at St-Simon Sud than at St-Simon Nord (Fig. 3). Assessment of the progressive development of the fouling community on the "Never" bags indicated that the major increase in fouling biomass occurred from mid-August to mid-October. As expected, the bags in the "Once" and "Never" handling treatments exhibited higher levels of biomass accumulation than either the "Biweekly" or "Monthly" treatments (Fig. 4). Note that the level of fouling on the "Never" turned bags was close to an order of magnitude higher than those handled on a "Bi-weekly" or "Monthly" basis.

3.2. Recruitment patterns on the oyster bags

Estimates of barnacle settlement based on the "Bi-weekly" plates suggested that the major recruitment event occurred in July at St-Simon Sud but extended from early July through early August at St-Simon Nord (Fig. 5). A few recruits were observed in early October but the numbers were limited. This pattern was confirmed by examination of the "Never" plates sampled at monthly intervals at each site, which indicated no significant increase in the number of barnacles after mid-August. Estimates of barnacle abundance/bag in October ("Never" bags) were lower at St-Simon Sud (2721 ± 600) than at St-Simon Nord (4250 ± 600).

Monitoring of mussel settlement rates indicated a peak in recruitment in early July with higher levels at St-Simon Sud than at St-Simon Nord (Fig. 6). The cumulative plates (i.e. "Never" treatment) indicated a decrease in the abundance of mussels following the July recruitment peak at both sites, followed by a second colonization event in September at St-Simon Sud. It is possible that the high summer temperatures in late July–early August caused mortality in the first set of juvenile mussels. Estimates of total mussel recruitment/bag in October were substantially higher at St-Simon Sud (1590 ± 205) than at St-Simon Nord (122 ± 52).

The ANOVA indicated that settlement levels for barnacles and mussels were significantly different between sites and among handling treatments (Table 2). The "Bi-weekly" maintenance schedule

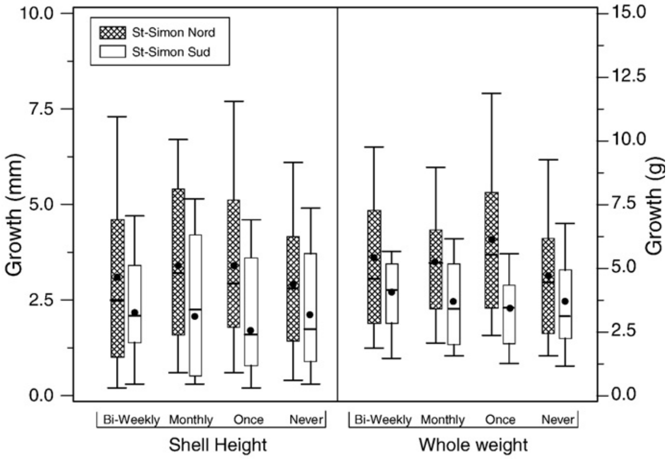

Fig. 9. Box plots of oyster growth in terms of shell height (mm) and whole weight (g) for the four handling frequencies at the two study sites (see Fig. 4 for further details on the box plot).

Table 4

A mixed model ANOVA indicating the effect of site (*S*), handling frequency (*H*), replicate treatment (*R*) and bag position within a line on oyster growth performance in terms of shell height (mm) and whole weight (*g*)

Source	DF	Shell height		Whole weight	
		MS	Pr>*F*	MS	Pr>*F*
Site (S)	1	150.2	<0.01	292	<0.01
Handling (H)	3	1.05	0.95	7.7	0.74
S×H	3	13.0	0.70	14.1	0.54
Rep (S H)	8	9.1	0.06	18.2	0.01
Bag (R S H)	32	4.2	0.90	6.2	0.55
Error	432	6.5		6.5	

yielded the lowest values of barnacles (125/bag) and mussels (49/bag) whereas the "Never" treatment yielded the highest values (3040 and 634/bag, respectively). The mean levels of barnacle and mussel abundance on the bags at the two sites for each treatment are shown in Fig. 7. The Site by Handling (SxH) interaction was not significant for the abundance of barnacles thus confirming the consistent impact of the handling treatment. This interaction term was, however, significant for mussel abundance; estimates were consistently higher for St-Simon Sud than for St-Simon Nord.

3.3. Recruitment patterns on the oysters

The number of mussels and barnacles attached to the oysters inside the bags was evaluated at the end of the study. In the case of barnacles, there were no significant differences between sites or among handling frequencies; the only significant effects were linked to the variance between replicates or among bags within a replicate (Table 3). This suggests that settlement intensity was patchy within the lease and/or among adjacent bags. Contrary to the pattern observed for the oyster bags, increased handling frequency did not reduce the recruitment rate of barnacles directly onto the oysters (Fig. 8). Estimates of barnacle abundance in October ranged from 10–30/oyster.

In the case of mussel settlement on the oysters, there was a significant difference between sites ($p=0.05$), and a significant effect of handling ($p=0.04$) (Table 3). Unlike the barnacles, mussel settlement patterns were similar across handling replicates or bags. Consistent with the observations of mussel recruitment on the oyster bags (Fig. 7), the mean abundance of mussels on the oysters in October was higher at St-Simon Sud than at St-Simon Nord (Fig. 8). Even though the "Never"-turned bags at St-Simon Sud had far more mussels attached to the outside (1200/bag) than all the other treatments, this was not reflected in the number of mussels attached to the oysters.

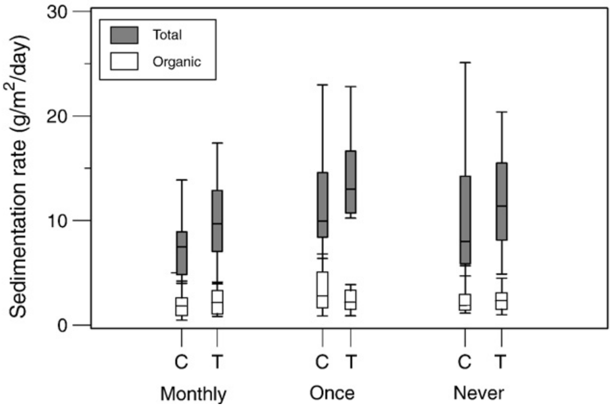

Fig. 10. Total and organic biodeposition rates (g dry wt m^{-2} d^{-1}) under the oyster bags in the "Monthly", "Once" and "Never" handling treatments vs. the reference sites prior to any handling activity. Note that the biodeposition rates under the "Never" bags were monitored four times over the season (see Fig. 4 for details on the boxplot). *T*=treatment sites with oyster bags, *C*=control or reference sites.

Table 5

A mixed model ANOVA testing for the effect of site (*S*), culture activity (*C*), handling frequency (*H*), time and trap replicate on total and organic biodeposition rates

Source	DF	Total biodeposition		Organic biodeposition	
		MS	Pr>*F*	MS	Pr>*F*
Site (S)	1	1552	<0.01	54.5	<0.01
Culture (C)	1	142	0.52	1.3	0.69
Handling (H)	2	142	0.52	4.0	0.62
Time (C H)	17	281	<0.01	8.3	<0.01
Trap	1	204	0.07	3.1	0.27
Error	137	61		2.53	

Culture activity was either present (experimental bags) or absent (reference sites).

3.4. Oyster growth performance

Growth estimates (shell height and whole weight) were obtained for the 10 labelled oysters deployed in the three oyster bags in each replicated handling treatment at each site (Fig. 9). The ANOVA revealed a significant Site ($p<0.01$) effect on shell height and a significant Site ($p<0.01$) and Replicate effect ($p=0.01$) on whole weight (Table 4). The frequency of Handling and the Bag had no significant effect on oyster performance. In general, the oysters exhibited very little growth over the duration of the study. In terms of shell height, an average increase of 3.2±0.2 mm was recorded at St-Simon Nord compared to 2.0±0.2 mm at St-Simon Sud. In terms of whole weight, an average increase of 5.9±0.2 g was recorded at St-Simon Nord compared to 4.3±0.2 g at St-Simon Sud. Although more frequent handling was effective in reducing biofouling levels, this effort did not translate into a significant increase in oyster growth. Note that oyster survival exceeded 95% in all treatments.

3.5. Biodeposition

The contribution of the fouling community and the oysters over the background biodeposition levels was determined by deploying sediment traps under the oyster bags for 24–48 h prior to any handling activity (Fig. 10). Biodeposition levels at the culture sites were not significantly different from those at the control or reference sites in terms of either total ($p=0.52$) or organic ($p=0.69$) material (Table 5). Mean estimates for the Reference sites were 12.3±0.9 g dry wt m^{-2} d^{-1} (total) and 2.7±0.2 g dry wt m^{-2} d^{-1} (organic) compared to 13.8±1.0 g dry wt m^{-2} d^{-1} (total) and 2.4±0.2 g dry wt m^{-2} d^{-1} (organic) at the Culture sites. Typical biodeposition levels under the bags handled "Monthly", "Once" or "Never" were also not significantly different in terms of total ($p=0.52$) or organic material ($p=0.62$). Significant temporal variation in total and organic biodeposition rates was observed ($p<0.01$) due primarily to increased wind activity in September–October. Levels of biodeposition were also significantly different between sites; mean values were higher for St-Simon Sud (16.6±3.4 g dry wt m^{-2} d^{-1}) than for St-Simon Nord (10.1±3.5 g dry wt m^{-2} d^{-1}). Note, however, that the mean organic content was not significantly different between the two sites: 20% at St-Simon Sud versus 21% at St-Simon Nord. No significant trap replicate effect was detected.

Biodeposition levels in the 48 h period following bag handling were compared to those at the control or reference sites using a mixed model ANOVA. Estimates of total biodeposition at the culture sites (113±13 g dry wt m^{-2} d^{-1}) were not significantly different ($p=0.65$) than those at the reference sites (145±13 g dry wt m^{-2} d^{-1}). The effect of handling frequency on biodeposition rates was highly significant ($p=0.01$). However, it should be noted that the final October handling of the "Never"-turned bags coincided with a major storm which produced values in excess of 400 g dry wt m^{-2} d^{-1} (total) at the reference sites and values of 300 to 400 g dry wt m^{-2} d^{-1} (total) at the culture sites. The occurrence of this environmental anomaly in conjunction with the unbalanced design (i.e. only one “Never” estimate vs. multiple “Monthly” estimates) effectively skewed the results of the handling frequency analysis.

3.6. Environmental monitoring

Temperature profiles were similar for the St-Simon Nord and St-Simon Sud sites (*t*-test on degree days, $p=0.57$), suggesting that these two adjacent bays experience similar conditions. Temperature levels above 20 °C were observed from mid-June to mid-September. No statistical difference was detected in chlorophyll levels ($p=0.95$) between the two experimental sites. The mean chlorophyll was 3.1 ± 0.6 µg L^{-1} with values ranging from 1.0 to 9.0 µg L^{-1}.

4. Discussion

The development of biofouling management strategies requires site-specific information on rates of biomass accumulation, the effectiveness of various husbandry techniques and the impact of these techniques on shellfish performance. While overall fouling biomass levels remained relatively low until mid-August, there was an exponential weight increase from September onwards. Comparison of various handling frequencies indicated that “Bi-weekly” and “Monthly” bag turning schedules were effective at reducing overall fouling levels, but bags turned “Once” in mid-August had similar low levels at the end of the season. Given the neutral effect of bag turning on oyster productivity, turning once in mid-August would appear to be an appropriate cost-effective strategy.

The two fouling organisms of greatest concern to oyster growers in the St-Simon area are barnacles (*Balanus improvisus*) and mussels (*Mytilus edulis*). Although both species are potential competitors for food resources, the major issue is the potential for increased labour costs. For example, barnacles must be hand-scraped from the oysters prior to marketing, whereas mussels bind oysters into clumps causing issues with growth, survival and retrieval from the bag. Comparing the “Biweekly” and cumulative monitoring results revealed that the barnacle settlement occurred primarily in July after which there was no further recruitment on the bags. One handling event in mid-August was sufficient to eliminate the majority of this population on the bag surface. In contrast the major peak in mussel recruitment occurred in July but there was ongoing settlement at a low level during late August–September. This observation suggests that bags may need to be handled in early October to eliminate any recently-settled juvenile mussels. Rapid growth of these mussels during the following spring and summer would likely lead to major handling issues.

One problem which was not solved by handling the bags was the unusually heavy settlement of barnacles directly on the oysters (20 barnacles/oyster). Research on the impact of fouling organisms which settle directly on cultivated shellfish species has typically revealed negative or neutral effects on productivity. Studies on pearl oysters (*Pinctada* spp) indicated that fouling of the shell by barnacles, oysters and other molluscs reduced growth and survival performance (Alagarswami and Chellam, 1976; Doroudi, 1994) and increased the incidence of shell deformities (Taylor et al., 1997). Likewise, the accumulation of fouling organisms on the shells of the tropical scallop (*Euvola (Pecten) ziczac)* had a negative impact on growth and survival (Lodeiros and Himmelman, 1996, 2000). In contrast Japanese oyster growers believe that certain types of fouling may benefit the growth of cultured oysters (*Crassostrea gigas*) (Arakawa, 1990). Specifically, it was hypothesized that the encroachment of encrusting forms such as sea squirts and sponges may “stimulate” shell growth whereas block forms such as barnacles and mussels have neutral or negative effects. Mazouni et al. (2001) suggested that the development of an ascidian fouling community on rope-grown oysters (*C. gigas*) in southern France may have positive implications for growth performance. They suggested that oysters benefit from the faeces and pseudofaeces of fouling ascidians which contain aggregates of picoplankton and other small particles not directly accessible to the oysters. Studies on the effect of fouling organisms attached to cultured mussels tend to suggest neutral effects on growth performance (Beristain and Malouf, 1988; Lesser et al., 1992; LeBlanc et al., 2003). The results of the present study did not suggest any direct negative or positive impact of the barnacle/mussel settlement on oyster performance.

One objective of this study was to determine whether an increased frequency of bag turning, i.e. reduced biofouling levels, would positively impact oyster growth performance. From mid-June to mid-October, average shell growth at both sites was less than 3.5 mm and none of the handling regimes was shown to promote oyster production relative to the “Never”-turned bags. Anecdotal evidence suggests that near market-size oysters exhibit their best growth in the late spring (May 1 to June 15) in northern New Brunswick (Hardy, unpubl. data); thereafter energy is likely diverted to gametogenesis and somatic tissue growth declines (Griffiths and Griffiths, 1987). Higher overall growth rates would likely have been recorded given a May 1 setup date, but the effect of bag handling on oyster performance would have been similar as most of the fouling occurs from mid-July onwards.

Determining the best husbandry practices for dealing with biofouling at a commercial scale requires specific information from that operation. Cultured shellfish species may vary widely in their response to the accumulation of biofouling as well as their sensitivity to handling. This complexity is reflected in the range of observations concerning the impact of fouling accumulation on shellfish performance. For example, various studies have indicated that the growth and survival of pearl oysters (*Pinctada* spp.) is negatively impacted by fouling of grow-out structures (Alagarswami and Chellam, 1976; Mohammad, 1976; Doroudi, 1994; Taylor et al., 1997). Regular cleaning was consistently linked to improved pearl oyster performance with no deleterious effects on survival (Taylor et al., 1997; Southgate and Beer, 1997, 2000). In the case of scallop species, fouling of pearl nets negatively impacted the growth performance of the tropical scallop (*Euvola (Pecten) ziczac)* (Lodeiros and Himmelman, 1996, 2000). Severe losses of cultured bay scallops (*Argopecten purpuratus*) in Chile were attributed to an infestation by the ascidian *Ciona intestinalis* (Uribe and Etchepare, 2002). In terms of fouling control, regular net changing resulted in a 68% higher muscle mass for sea scallops (*Placopecten magellanicus*) (Claereboudt et al., 1994a), and improved growth in the great scallop (*Pecten maximus)* with no decrease in survival (Louro et al., 2007). In contrast, other studies have not detected any impact of biofouling on the growth of bivalves reared in enclosed structures (Wallace and Reisnes, 1985; Widman and Rhodes, 1991, Lodeiros et al., 1993). Certain researchers have even suggested the possibility of a positive interaction between the fouling community and the cultured species (Mook, 1981; Ross et al., 2002). In some cases, frequent maintenance activity has been observed to negatively impact cultured bivalve performance. Handling bay scallops (*Argopecten irradians irradians*) on a monthly basis was detrimental to growth and survival relative to those handled less often (Widman and Cooper, 1996). Likewise, more frequent manual handling of juvenile sea scallops (*P. magellanicus*) for the purpose of grading and net changing was found to negatively impact survival (Parsons and Dadswell, 1992). Although pearl oyster growth was significantly better in trays cleaned every 4 or 8 weeks rather than 16 weeks, oysters handled every 4 weeks had lower survival rates (Pit and Southgate, 2003). In the present study there was no indication of lower oyster survival in the bags that were turned more frequently—survival exceeded 95% in all treatments. However, observations suggested that bag turning may damage the delicate fringe associated with new shell growth. Hence more frequent handling may effectively negate any positive impacts associated with improved flow rates and/or an increased food supply.

In the present study, the overall mean level of biodeposition under the floating bags, including the major storm event in October, (40 g dry wt m^{-2} d^{-1}) was similar to that at the reference sites (36 g dry wt m^{-2} d^{-1}). These levels were comparable to those recorded under floating bags (50 g dry wt m^{-2} d^{-1}) in St-Simon Sud from June 2002 to October 2003 (Mallet et al., 2006). Despite the accumulation of fouling organisms on the “Never” turned bags, there was no significant increase in biodeposition levels relative to the reference sites. The observation of a significant site effect was consistent with Crawford et al. (2003) who noted that the variation in biodeposition levels among three oyster culture sites in Tasmania was greater than that observed between the culture and reference locations within a site. The highest biodeposition values recorded overall (475 g dry wt m^{-2} d^{-1}) were from the reference sites in St-Simon Nord following a massive fall storm in October 2006. Note that a similar storm event resulted in unusually high sedimentation levels (240 g dry wt m^{-2} d^{-1}) in St-Simon Sud in late September 2003 (Mallet et al., 2006). It is likely that these fall storm events effectively re-suspend and disperse sediment which may have accumulated over the summer months.

It should be noted that the typical biodeposition rates in the present study ($<$20 g dry wt m^{-2} d^{-1}) are considerably lower than estimates for biodeposition under mussel rafts in Spain (515 g dry wt m^{-2} d^{-1}) (Perez-Camacho et al., 1991) or mussel longlines in Prince Edward Island (40 to 60 g dry wt m^{-2} d^{-1})(Grant et al., 2005). This difference is related in part to the lower density of culture as well as the containment of the inventory. Unlike mussel culture there is no loss of individuals to the bottom; biodeposition is associated primarily with the metabolic activities of the oysters and the fouling community. Even under the “Never”-turned floating bags, either pre- or post-handling, levels of total and organic biodeposition were not significantly higher than at the reference sites.

5. Summary

In conclusion, the accumulation of fouling biomass on floating bags containing cultured oysters can be effectively controlled by means of bag turning. This was evident both in terms of the reduction in the total amount of fouling as well as the decline in the number of barnacles and mussels. Turning the bags only once in mid-August reduced overall fouling levels substantially and decreased the abundance of barnacles to near negligible levels for the remainder of the grow-out season. A second handling, however, may be necessary in the fall to eliminate the late mussel set prior to winter. Although more frequent handling (i.e. “Bi-weekly” or “Monthly”) consistently resulted in lower overall fouling levels, there was no advantage in terms of enhanced oyster growth performance or reduced biodeposition rates. In general, oyster growth performance was poor at both sites ($<$1 mm/mo) and was not enhanced by improving access to food resources. Observations suggested that the activity of bag turning may be detrimental to growth by damaging the delicate shell edge. Given the relatively short growing season in northern New Brunswick and the possibility that bag turning may be detrimental, the frequency of handling should be restricted to once or twice during the season and efforts directed towards developing alternative strategies for keeping the bags/oysters clean. Overall, there was no indication that floating bag oyster culture, even in cases where the bags were heavily fouled, significantly increased biodeposition levels relative to the reference sites.

Acknowledgements

Funding for this work was provided by the Aquaculture Collaborative Research and Development Program of the Department of Fisheries and Oceans and the oyster company L'Étang Ruisseau Bar Ltd. L'Étang Ruisseau Bar Ltd also provided technical support and equipment to undertake this research. Special thanks are extended to Julie Mallet and Janie Fauteux who participated in the processing of the samples as well as to Marc Skinner for assistance in the field. We would also like to acknowledge the constructive comments of several anonymous reviewers.

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STANDARD OPERATING PROCEDURES FOR THE ENVIRONMENTAL MONITORING OF MARINE AQUACULTURE IN NOVA SCOTIA

NOVA SCOTIA
Fisheries and Aquaculture

June 2020

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Standard Operating Procedures for Environmental Monitoring of Marine Aquaculture Sites in Nova Scotia

1 INTRODUCTION

The *Standard Operating Procedures for Environmental Monitoring of Marine Aquaculture Sites in Nova Scotia*, describes the monitoring and laboratory methodologies for the Nova Scotia Environmental Monitoring Program (EMP). Both marine finfish and marine shellfish farms in NS are required by the Nova Scotia Department of Fisheries and Aquaculture (NSDFA) to comply with the EMP as outlined in the Aquaculture Management Regulations under authority of the *Fisheries and Coastal Resources Act*. Provided in this document are monitoring instructions, laboratory guides, field templates and reporting requirements designed to assist those conducting environmental monitoring on a marine aquaculture lease. This document and methodologies described within will be reviewed yearly to include changes and innovations to field methods, laboratory techniques, technologies and regulatory approaches.

This EMP Standard Operating Procedure (SOP) originated in 2002 as part of the document titled, *Design of the Environmental Monitoring Program for the Marine Aquaculture Industry in Nova Scotia* (Smith et al., 2002). Several revisions have been made to the EMP SOPs and framework; these revisions incorporate the latest advancements in science and technology. This helps to ensure that the EMP is up-to-date, relevant, and effective. The EMP is a mandatory requirement, and integral part of the leasing and licensing process. Marine finfish and shellfish farm operators are responsible to adhere to this program, coordinate monitoring as instructed and provide results to NSDFA as required.

Should readers of this document have any questions, please contact the Manager of Aquaculture Operations at (902) 875-7434.

2 DETERMINATION OF MONITORING STATIONS

This section provides guidance on determining the number and position of monitoring stations required for EMP. The following criteria are to be considered in making these determinations:

- Level of the monitoring event being conducted (Level I, II or III)
- Maximum number of fish onsite during the current production cycle (Table 1)
- Prevailing current direction relative to the shoreline (Table 2)
- Biomass contained within each cage at the time of sampling (Figure 1)
- Water depth at cage edge
- Bottom type and site conditions
- Historical environmental performance

Site-specific conditions may prevent the positioning of monitoring stations exactly as described in this SOP. If the operator or third-party operator is aware of conditions that may prevent a station from being located in the correct position, they must notify NSDFA and receive approval for any deviations from the SOP prior to the sampling event. Any deviations from the SOP that could not be pre-determined do not have to be approved by NSDFA but must be submitted in the final report.

2.1 Pre- Monitoring Submissions

At a minimum of two weeks prior to an anticipated monitoring event, the operator or the third-party operator are required to submit the following information to NSDFA for review:

- A detailed site diagram or aerial image indicating:
    - Biomass contained in each cage, in kilograms
    - Proposed location of all monitoring stations
    - Proposed alternative sampling locations for all stations located on cage-edge
    - Location of any assigned Historic High stations (if applicable)
    - Location of Reference Station to be sampled
- Anticipated monitoring date
- Monitoring Equipment that will be used, including:
    - Sediment sampler (see Appendix A2)
    - Video camera system
- Details regarding any requested deviations from the sampling methods specified by this SOP

2.2 Number of Finfish Monitoring Stations

The minimum number of monitoring stations required for each finfish aquaculture lease is based on the maximum number of fish on site during the current production (Table 1). A minimum of two monitoring stations are required for sites containing a maximum of 1-200,000 finfish. The number of required monitoring stations will increase by one for every additional 100,000 finfish stocked.

If more than one cage array is found within the same lease, each array will be treated individually. For example, if one lease has a maximum of 700,000 fish, and the first array contains 250,000 fish and the second array contains 450,000 fish. The first array would require three monitoring stations and the second array would require five monitoring stations.

In addition to the monitoring stations specified in Table 1, historic high monitoring stations must also be sampled as part of the EMP. Historic high monitoring stations are those soft bottom monitoring stations whose mean sulfide levels have previously exceeded 3000 µM. These stations must be re-sampled annually, until the mean sulfide level for that station decreases below 1500 µM. Historic high stations must be located within 10 m of the original sampling coordinate. If samples are collected at a distance greater than 10 m from the original coordinates, the results will not be considered valid for determining the recovery status of the station. In cases, where multiple historic high stations are located within 10 m of one another, NSDFA may consider reducing the number of stations required for re-sampling upon request.

For sites that are inactive at the time of the anticipated monitoring event, operators or third-party operators, should consult the *Policy for Sampling Inactive Sites* to determine the appropriate requirements.

Table 1. Number of Monitoring Stations Required for Level I Sediment and Video Collection

Maximum number of fish within cage site array during production cycle	Number of sampling stations (not including reference stations)	Number of samples (3 samples/station for soft bottom sites)
1-200,000	2	6
200,001-300,000	3	9
300,001-400,000	4	12
400,001-500,000	5	15
500,001-600,000	6	18
600,001-700,000	7	21
700,001-800,000	8	24
800,001-900,000	9	27
900,000-1,000,000	10	30

*Contact NSDFA if more than 1,000,000 finfish are stocked and when number of sampling stations exceeds number of cages

2.3 Position of Finfish Monitoring Stations

The position of monitoring stations for Level I and Level III EMP will be determined using the following criteria:

- position of the cage array relative to the shoreline
- direction of the prevailing water current
- current speed
- cage biomass
- water depth at cage edge
- bottom type
- site conditions

The application of these criteria in selecting sampling locations is further outlined in Table 2. Examples of prioritized selection of sampling locations are provided for sites with generally linear current flow (Figure 1) and for those with generally curving flow (Figure 2). All such monitoring stations will be located at a cage edge, along the outside perimeter of the array. Where multiple sediment samples are to be collected, samples must be taken from three separate locations along the outer perimeter of the cage (Figure 3). The samples must be collected at a distance far enough apart to ensure that samples are not taken from a location that was disturbed by a previous sampling attempt.

Accurate recording of monitoring station locations is vital for the efficacy of the EMP and helps to ensure the consistency and repeatability of a monitoring event. As such, vessels are required to be moored to cages while conducting monitoring activities associated with a cage edge. Mooring is not required for stations that are not located at cage edge (e.g. historic high and reference stations). However, an appropriate method to remain within 10m of the assigned station coordinates must be employed. If surface deployed monitoring equipment is being used to sample a cage edge monitoring station, this equipment must be deployed no more than 3 m away from cage edge. A GPS waypoint must be recorded at every monitoring station using the **NAD83 CSRS** datum and is to be submitted to NSDFA in decimal degrees.

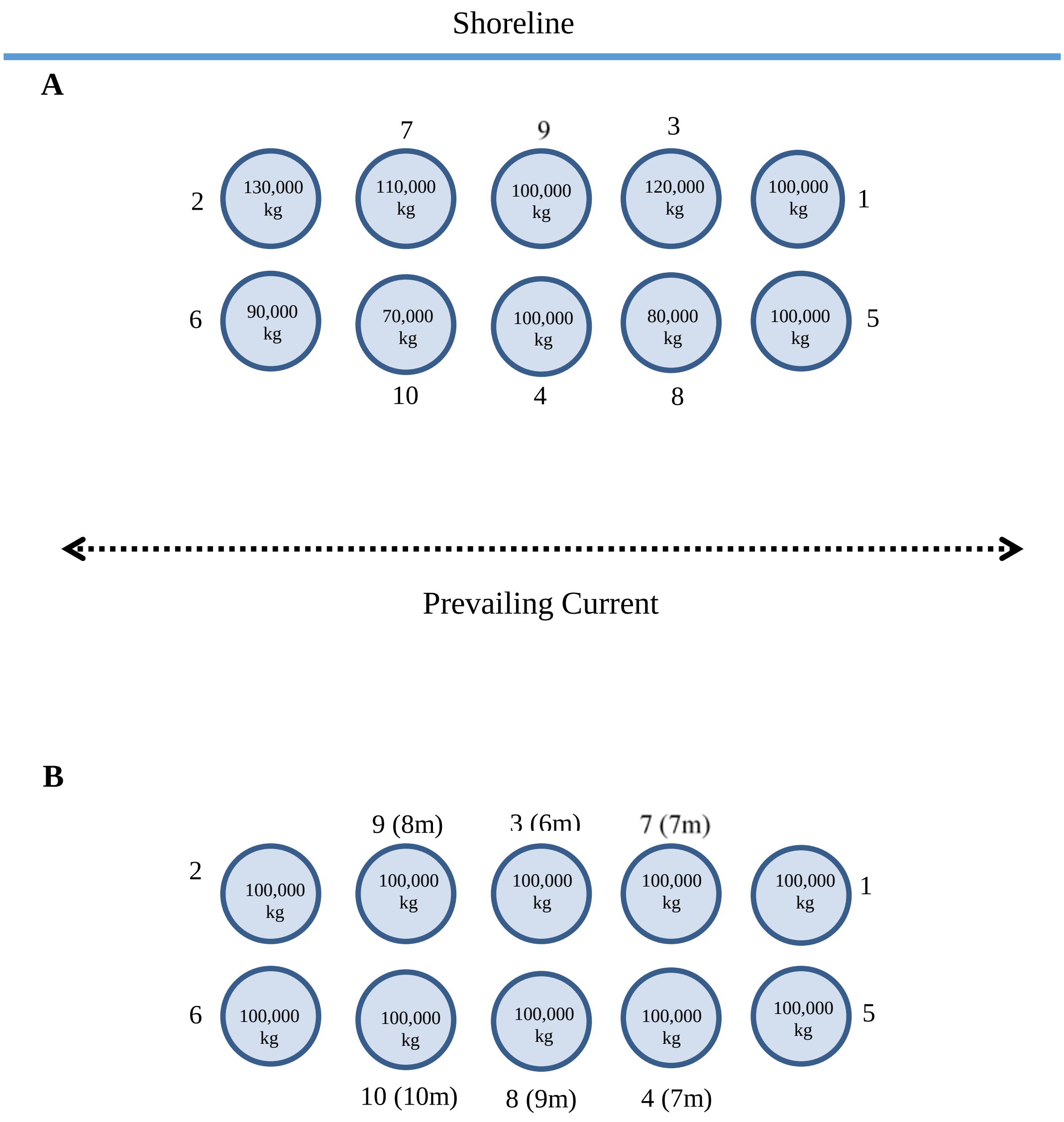
When sediment samples are collected using a surface deployed grab sampler, the depth of the station must be recorded. The station water depth can be recorded using either a weighted drop line or equipment found on the vessel. If a weighted drop line is used, it should be deployed after sample collection is complete.

Where samples are collected by a scuba diver, a weighted drop line will be used to assist in locating the sampling location on the seafloor and the DGPS coordinates must be recorded. Care must be taken to ensure sample locations have not been disturbed by the impact of the drop line anchor on the seafloor. All required samples will be collected in similar substrate within 1 m of the drop line anchor. If sediment cannot be retrieved from this area, divers may move to the closest undisturbed sediment for sample collection. Such deviation must be noted in the report along with an estimate of distance from the drop line anchor.

In situations where site infrastructure or other obstructions prevent access to a proposed monitoring station location, a revised sampling location should be established. The revised monitoring station must be located as close to the cage with the highest biomass, without risking entanglement of equipment. As with any other monitoring station, a GPS waypoint (using **NAD83 CSRS** datum) must be logged at the new location. If a reference or historic high station cannot be monitored, then the operator or third-party operator must record the distance and direction of the revised station from the target sampling waypoint. If a monitoring station location is revised, coordinates of the new sampling location and an explanation of the spatial variation must be provided in the final report.

Table 4. Position of Monitoring Stations for Level I and Level III Sampling

Current Type	**Monitoring Station Positions**
Generally linear (uni- or bidirectional) with moderate or high current speeds	**1)** Cages along the outside perimeter of each array will be selected for positioning of monitoring stations. **2)** Stations 1 and 2 will be positioned at opposite ends of the array, in alignment with the prevailing water current pattern and on the cages nearest the shoreline. If identified cage is empty, station is positioned at edge of next stocked cage closest to shoreline at time of monitoring. **3)** Station 3, if required, will be positioned approximately at right angles to the prevailing water current on the shore side of the array on the edge of cage with the highest biomass at time of monitoring. For this station and all additional, if biomass of two or more cages is equal, station is to be positioned on edge of cage with shallowest water depth. **4)** Station 4, if required, will be positioned approximately at right angles to the prevailing water current on the side opposite to station 3 on the edge of cage with the highest biomass at time of monitoring. **5**) Station 5, if required, will be positioned on the same side as station 1 on the edge of cage with the next highest biomass at time of monitoring. **6)** Station 6, if required, will be positioned on the same side as station 2 on the edge of cage with the next highest biomass at time of monitoring. **7)** Stations 7 and 8 on the same sides as stations 3 and 4 respectively on edge of cage with next highest biomass at time of monitoring. **See Figure 1 for visual representation**
Generally curving (eddies, or follows shoreline, uni or multidirectional) or low current speeds	**1)** Cages along the outside perimeter of each array will be selected for positioning of monitoring stations. **2)** Stations 1 and 2 will be positioned approximately at right angles to each other, with one on the shore side and the other aligned on the edge of cage nearest to shore with the highest biomass at time of monitoring. For these stations and all additional, if biomass of two or more cages is equal, monitoring station is to be positioned on edge of cage with shallowest water depth. **3)** Station 3, if required, will be positioned on the side opposite to station 2 on the edge of cage nearest to shore. If empty, station is to be placed on edge of next closest cage to shore on that side. **4)** Station 4 if required, will be positioned on the side opposite to Station 1 on the edge of cage with the highest biomass. **5**) Station 5, if required, will be positioned on the same side as station 1 on the edge of cage with the next highest biomass. **6)** Station 6, if required, will be positioned on the same side as station 2 on the edge of cage with the next highest biomass. **7)** Station 7, if required will be placed on the same side as station 4 on the edge of cage with the next highest biomass. **8)** Station 8, if required will be placed on the same side as station 3 on the edge of cage with the next highest biomass. **See Figure 2 for visual representation**



Legend:

Cage with biomass: 100,000 kg

Sampling station: 1-10

Station depth: (XXm)

Figure 1. Examples of monitoring station positioning at sites with generally linear water current patterns, where arrays contain varying biomass per cage (A) or equal biomass distribution (B).

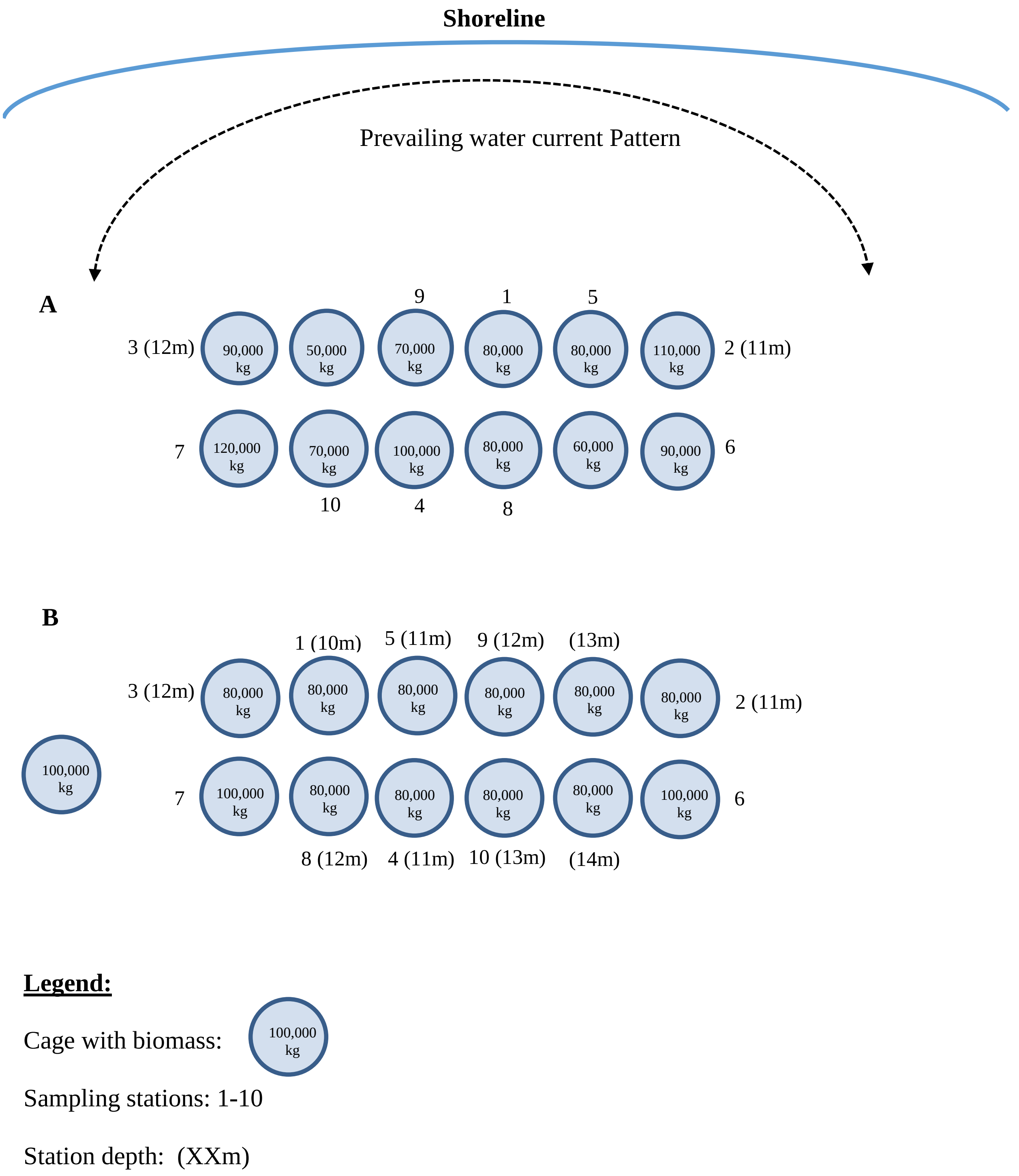


Figure 2. Examples of monitoring station positioning at sites with generally curving water current patterns, where arrays contain varying biomass per cage (A) or equal biomass distribution (B)

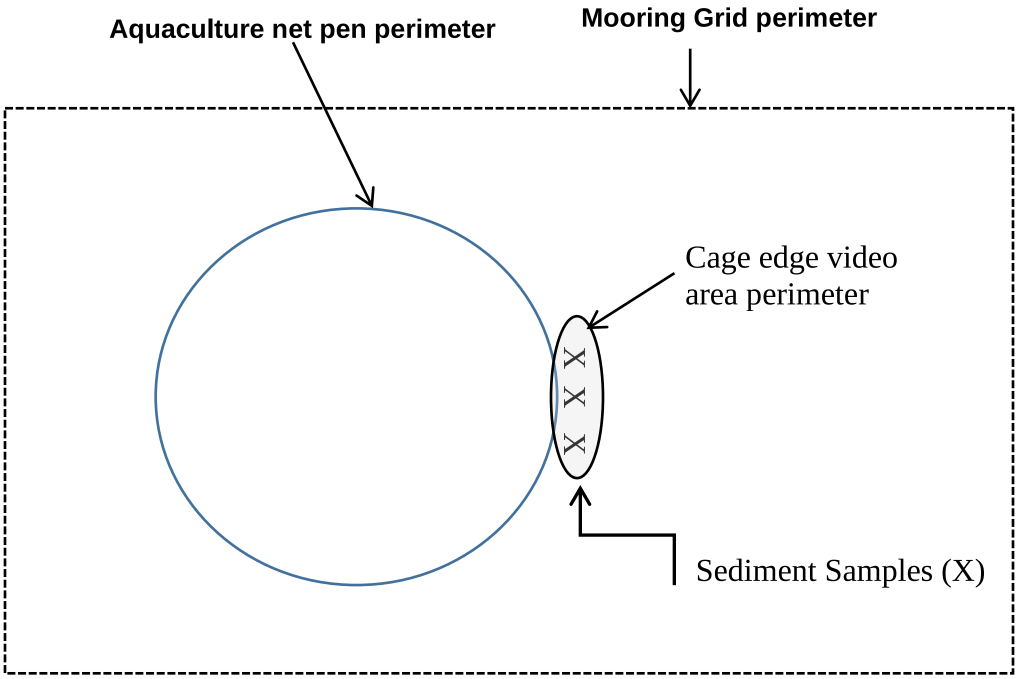


Figure 3. Position of sediment samples and video monitoring for soft bottom stations for Level I, II, and III monitoring events.

2.4 Shellfish Monitoring Stations

Environmental monitoring for shellfish farms will be scaled to the level of risk associated with the operation and will consider the following: production level, percent of bay volume, and, historical environmental performance. Refer to Figure 2: Risk Based Decision Making Matrix of *Environmental Monitoring Program Framework for Marine Aquaculture in Nova Scotia* for elaboration on appropriate monitoring actions (PNS, 2020).

Alternative levels of monitoring may be proposed for shellfish aquaculture sites that have repeatedly shown no or limited potential for impact. These may include reduced monitoring requirements, video monitoring only, or monitoring at extended spatial and temporal intervals. Shellfish farms that have no production will not require environmental monitoring.

2.5 Reference Stations

Each marine aquaculture lease undergoing environmental monitoring requires that a minimum of one reference station be sampled. Reference stations are established during baseline sampling. Reference stations must be located between 100 and 300 meters from the lease boundary, in the direction of the dominant current. Reference stations must be positioned in an area with a similar depth and sediment type to what is found at stations sampled within the lease boundary. If the required distance criterion cannot be achieved, reference stations should be positioned in an area with similar characteristics to the monitoring stations within the lease boundary (water depths and sediment type, etc.).

If acceptable sediment samples cannot be collected at a previously established reference station, a new reference station should be established. A new reference station can only be established after a minimum of 5 unsuccessful attempts are made to collect sediment at the original reference station. A new reference station must meet the distance, depth and sediment type criteria detailed above and the new coordinate must be submitted in the final report to NSDFA. If a new, soft-bottom, reference station cannot be established which meets these criteria, a 200-meter video transect, starting approximately 100 meters from the lease boundary and ending approximately 300 meters from the lease boundary will be conducted. Video collection is to be conducted as described in Section 3, with drop-camera video stations located at 50-meter intervals.

2.6 Monitoring Levels

Level I EMP events are conducted annually, between July 1 and October 31, and are the primary means of monitoring conducted at active aquaculture sites in Nova Scotia. Determination of the positioning and number of required stations for Level I monitoring is outlined in Sections 2.1 to 2.5.

Level II monitoring events are required when the results of annual Level I monitoring classify a lease as Hypoxic B, Anoxic, or having failed based on the mixed or hard bottom classification protocol. In such cases, a consistent rationale for additional monitoring will be applied based on the following monitoring objectives:

a) Improving spatial delineation of the impacted area. This will include the establishment of cage-edge monitoring stations at all cages immediately adjacent to Level I monitoring stations with mean sulfide concentrations $\geq$ 3000 µM or which failed to meet the Environmental Quality Objectives (EQO) for hard bottom stations (Framework Section 2.2.3.3)

b) Improving spatial delineation of the zone of influence. Monitoring stations will be established at the four corner compensator buoys of the array as well as additional perimeter compensator buoys at no more than 200m spacing along the outer edge of the array. If compensator buoys are not utilized as part of the system design, contact NSDFA for sampling guidance.

Level II monitoring events do not require the inclusion of a reference or historic high stations.

A site will be initially classified using the results from the Level I sampling event. If Level II monitoring is required for a site, then the final site classification will be based on the results from this monitoring event. The classification of the site will dictate the most appropriate site management responses for each aquaculture site. These site management responses can include things such as follow up monitoring and/or the implementation of mitigation measures. Please see Section 3.0 of the *Environmental Monitoring Program Framework for Marine Aquaculture in Nova Scotia* for more detail on classification of sites and management responses (PNS, 2020).

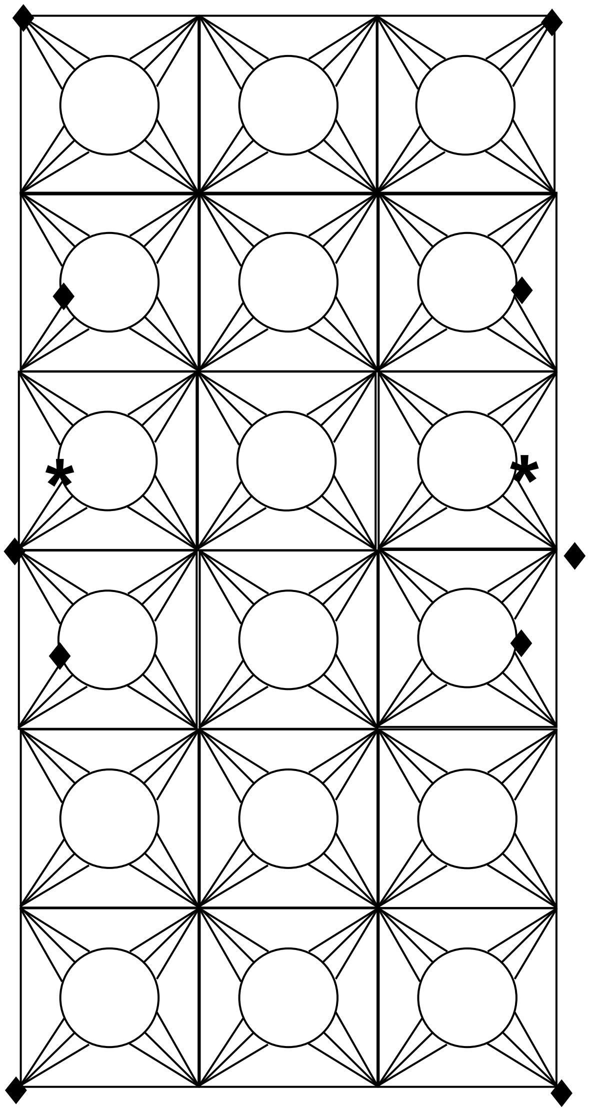


Figure 5. Example of Level II monitoring station placement (diamonds) relative to stations where average free sulfide concentrations were found to be ≥3000 µM (stars).

Level III monitoring is required when a site consistently fails to meet oxic conditions or when the results of annual Level I monitoring classify a lease as Anoxic or otherwise severely impacted. This sampling is used to capture seasonal variation on a lease and is used to closely monitor affected areas within the lease boundaries through increased temporal sampling intensity. Level III sampling events will take place between March 1 and May 31 of the year following the triggering results. Monitoring will target all sampling stations visited during the previous Level I monitoring event and may also include additional requirements as determined by NSDFA in discussion with the site operator.

If Level III monitoring is required on an aquaculture lease, the results from both Level I and Level II sampling will be evaluated to determine the state of the benthic environment within the lease. The results will be used by regulators to provide site-specific recommendations for any remedial action that is required by the operator.

2.7 Timing of Monitoring

All Level I and Level II sampling events must be completed annually between July 1st and October 31st. If Level III sampling is necessary for a site it must be completed between March 1st and May 31st of the following year. All attempts must be made to complete any sampling event in a single day. If it is anticipated that more time will be required to complete the sampling event a request may be made for an approved deviation. In such cases, a maximum of two consecutive days will be allowed for completion of sampling.

A sampling extension may also be granted if unavoidable circumstances or equipment malfunction prevent an ongoing sampling event from being completed in the approved sampling timeframe. If an extension is required to complete the sampling event, NSDFA must be consulted to request an approved deviation. If the remaining sampling can not be completed within a five-day period, the results of this sampling event will not be accepted by NSDFA and the entire sampling event will have to be repeated.

Extensions will not be granted as a result of inclement weather. Those conducting required monitoring must plan appropriately to ensure that weather will not prevent sampling from being completed within the prescribed timeframes.

3 BENTHIC VIDEO COLLECTION

3.1 Video Recording Methodology

Benthic video footage must be collected at every monitoring station during all levels of monitoring. Video may be recorded via surface deployed drop camera, hand-held diver operated video camera or remotely operated vehicle and must be collected prior to sediment sample collection. The criteria for acceptable video recording are described below:

- A placard containing relevant video station details (date, time, coordinates, lease number and station ID) must be presented at the beginning of each video recording**,** prior to submersion.
- A 360º panorama (or as close as possible) of the water surface view plane must be recorded at each video station prior to submersion.
- The video must include continuous footage of the initial descent, impact with the seafloor, camera ascent and retrieval on deck.
- The field of view must include a visible reference scale. If measurements are not indicated on the reference scale, the measurements must be submitted to NSDFA as part of the video submission.
- Surface deployed camera video must include a digital overlay detailing real time latitude and longitude of the monitoring station. The latitude and longitude should be formatted using the NAD83 CSRS datum and submitted in decimal degrees.
- Hand-held, diver collected video must include a view of the current coordinate location on a sufficiently accurate DGPS unit both before and after entering the water.
- Once near bottom, the camera’s decent will halt above the seafloor. Demonstration of benthic consistency will then take place via camera or diver contact with the sediment.
- Video lighting and resolution must be sufficient to allow for the characterization of sediment conditions, identification of macro flora and fauna and accurate interpretation of the presented reference scale.
- A minimum 2 minutes of seafloor footage are required at each video station.
- Video for each monitoring station must cover a minimum area of 5 m^2.

If any video submitted to NSDFA does not meet the video quality criteria listed above it may not be accepted.

3.2 Video Station Locations

3.2.1 Soft Bottom Monitoring Stations

At monitoring stations that are determined to be Soft Bottom, (Framework Section 2.2.1) a single video station will be completed at the identified cage edge or within 10m of the assigned target coordinate. Recording at such video stations are to be carried out as specified in Section 3.1.

3.2.2 Hard Bottom Monitoring Stations

At any monitoring station that is determined to be Hard Bottom, (Framework Section 2.2.1) video recording is to be conducted in accordance with the Department of Fisheries and Oceans Canada (DFO) *Aquaculture Activities Regulations* and the following criteria:

- Recordings will be conducted along a transect extending away from the monitoring station in a direction perpendicular to the edge of the cage array on which the monitoring station lies.
- A total of six video stations will be established along the length of the transect, at 0 m, 10 m, 20 m, 30 m, 40 m. and 50 m from cage edge.
- Recordings at all video stations are required to meet the methodology and quality criteria outlined in Section 3.1 unless otherwise directed.
- A 360º panorama is only required at the beginning of each individual recording (i.e. if the camera system is recording continuously throughout multiple video stations of a transect a panorama is only required prior to submersion at the first station.)
- If utilizing a diver-held or ROV mounted camera to conduct a continuous transect, a weighted drop line or other visual guide must be used to mark the transect line as well as the individual video stations at 10m increments.
- A diver or ROV conducting a continuous transect will do so at a speed and height above bottom which allows for the clear observation and identification of macro flora and fauna within 1 meter to either side of the established transect line

If any video submitted to the NSDFA does not meet the video quality criteria listed above it may not be accepted.

3.3 Video Observation Requirements

For each established video station visited during a monitoring event, detailed observations are to be made, recorded and submitted to NSDFA. Appendixes A3 and A4 are provided as sample templates for the recording of field observations. Observations at each video station should include, but are not limited to:

- Video Station Details:
    - Waterbody
    - Aquaculture lease number
    - Station ID
    - Distance along established transect (where applicable)
    - Date and time of recording commencement
    - Water depth
    - Latitude and longitude
    - Distance and direction from assigned station (where applicable)
- Video Observations:
    - Sediment Description
        - Colour at surface and subsurface
        - Composition (e.g. sand, cobble, boulder, etc.)
        - Consistency/Consolidation (e.g. soft, hard, easily disturbed etc.)
    - Benthos Description
        - Macrofauna observed
        - Macroflora observed
        - Presence and relative abundance of uneaten finfish feed
        - Presence and relative abundance of finfish faeces
        - Presence and relative abundance of other organic detritus
        - Presence of gas bubbles released from sediment
        - Presence and approximate % coverage of *Beggiatoa* like bacterial mats
        - Presence and approximate % coverage of polychaete complexes
        - Presence and approximate % coverage of barrenness
        - Anthropogenic debris observed
    - Biophysical conditions at depth
        - General visibility
        - Relative current speed
        - Relative abundance of suspended particulate matter

In the case of a continuous video transect collected by a diver or ROV, the above observations must be recorded at a minimum of 10 m intervals with note made of any significant changes occurring in the interim. Note should be made of the presence of any macroflora, macrofauna or significant environmental indicators observed at any point throughout the transect.

4 SEDIMENT COLLECTION

Samples of benthic sediment are required to be collected at each station during all levels of monitoring. The goal of sediment collection is to retrieve representative samples of the current benthic environment in order to assess relative levels of health and impact. Depending on the level of sampling, collected sediments will be analyzed for oxidation-reduction potential, total dissolved sulfide concentration, porosity, and sediment organic matter. Although sulfide concentration is the main regulatory indicator used to classify an aquaculture lease, the other variables are still important and are used to validate and confirm accuracy of sulfide results via the empirical relationships of measured variables (Hargrave, 2010) and the Benthic Enrichment Index (BEI) (Hargrave, 1994).

Sediment samples may be collected via surface deployed equipment (e.g. dredge, grab, gravity corer) or manually operated core tube (diver-held or ROV). Selection of the appropriate sampling method and equipment to be utilized at a station will depend largely on site-specific conditions such as sediment composition and consolidation, water depth and current speed. A decision tree is provided in **Appendix A2** to serve as guidance in the selection of appropriate equipment. Proposals for the use of alternate methods or equipment not listed in this SOP should be submitted to NSDFA prior to sampling for review and approval.

In all cases, sample collection must take place in a consistent and repeatable manner in order to maintain the integrity of the subsequent analysis results. All samples, regardless of collection method, are required to meet the following methodology and quality criteria in order to be considered acceptable:

- Triplicate samples are to be collected at all stations
- Triplicate samples are to be sub-sampled from discrete sediment collection events. (i.e. a single sub-sample collected from each of three grabs or three separate core tubes)
- Each collection will have a minimum sediment depth of 5 cm
- Sub-samples will be collected or directly analyzed from the top 2 cm of the collected sediment
- Overlying water must be present over the entire sample surface at time of retrieval
- The interface between the sediment surface and overlying water is relatively flat and undisturbed
- Sediment sampling equipment must not be overfilled
- All efforts must be made to collect sediment from seafloor that has not been disturbed as a result of previous sample or video collection
- Accurate GPS coordinates are to be recorded at the location of each sample collection
- Photographic record of the results of each sample collection attempt are to be collected prior to sub-sample extraction or analysis.

Additional, equipment-specific, quality criteria are provided in the following sections.

4.1 Surface Deployed Sampling

Surface deployed sediment collection equipment may be of the core, grab or dredge type and should be selected, based on site and environmental conditions, in order to best satisfy the methodology and quality criteria of this section.

- Grab samplers must have sufficient opening to access and observe the entire surface of a collected sample
- All sampling equipment is to be operated as per the manufacturer's specifications. Any modifications made must be approved by the NSDFA prior to use
- The speed at which the sampler descends through the water column must minimize the disturbance of benthic surface sediment due to the force of water being displaced
- Sediment collection equipment must descend and ascend vertically to ensure the sampler connects evenly with the seafloor and that the collected sample is not shifted during retrieval
- Retrieval of the sampler should start by slowly lifting from the seafloor and then steadily raising it to the surface at a target speed of 30 cm/s or less (Environment Canada 1994). Sample retrieval speed must be calculated and included within the submitted report to NSDFA
- If equipment uses covering flaps to protect samples during retrieval (e.g. Ponar or Van Veen grabs), flap position should remain down throughout deployment
- If an acceptable sediment sample has not been successfully collected after three consecutive attempts, uncontrolled descent (free-fall) of the sampling equipment will be permitted. In such cases, the sampler should be allowed to free fall for no more than a few meters above the bottom
- Sampler jaws must be fully closed upon retrieval (i.e. rocks or shells should not be holding bottom open)
- Collection attempts will only be considered unsuccessful if the failure is due to characteristics of the sediment composition or consolidation. Failure resulting from the presence of site debris or other anthropogenic factors will not count towards the number of unsuccessful attempts
- Overlying water will be removed via siphoning before sub-sampling occurs
- Photographs of the entire sample surface are to be collected before and after the removal of overlying water and before sub-sampling occurs
- Sub-samples should be collected using plastic syringe cores with a rubber-tipped plunger and mL increments (e.g. Becton-Dickson 5 mL, Fisher # 14-823-35) with the tapered tip removed
- Sub-samples collected from successful retrievals must be extracted from a minimum of three different locations within the sample
- Sediment remaining after subsampling must be discarded away from subsequent sampling locations
- Sampling equipment must be rinsed thoroughly between deployments
- If sediment consolidation or composition has resulted in five failed collection attempts before three acceptable samples can be retrieved, an alternate station is to be established at the cage on the same side of the array which contains the next highest biomass and which is not already the target of a sampling station

- If an alternate station can not be established which meets the above criteria, visual monitoring, as described in Section 3.2.2, will be conducted in lieu of sediment sampling at the initially selected location
- If sediment consolidation or composition has resulted in five failed collection attempts before three acceptable samples can be retrieved at the alternate location, the station will be considered a ‘hard bottom station’ for that sampling event. Visual monitoring, as described in Section 3.2.2, will be conducted in lieu of sediment collection

Any sample collection that fails to meet these methodology and quality criteria may not be accepted by NSDFA.

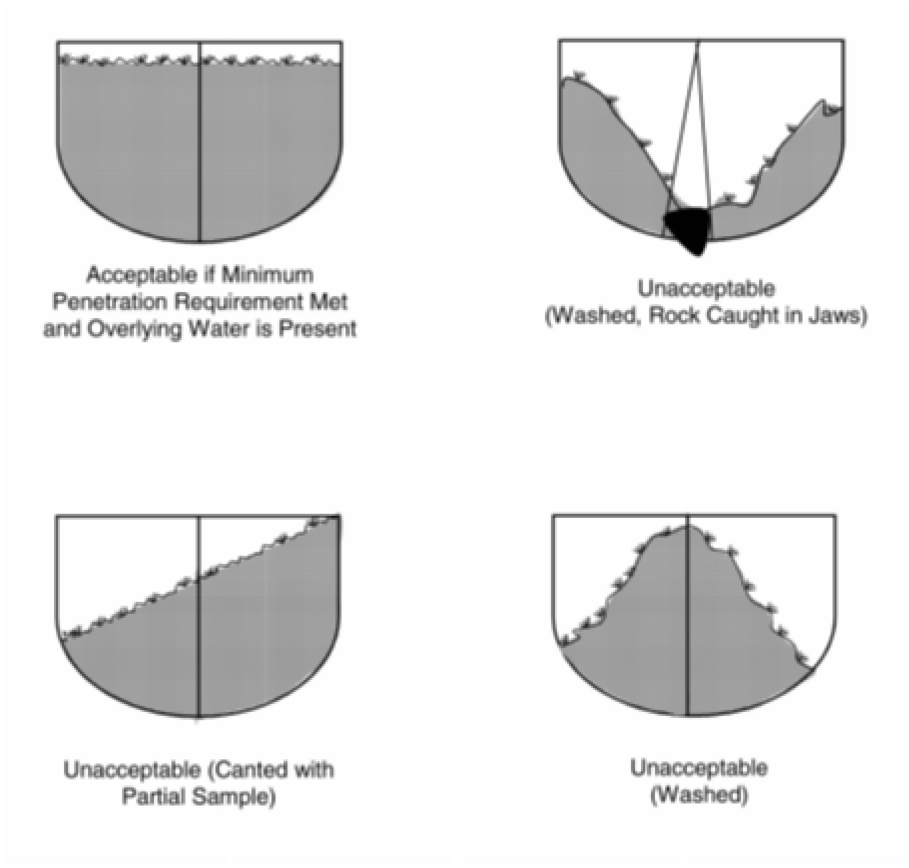


Figure 8. Illustrations of acceptable and unacceptable grab samples (USEPA 2001)

4.2 Sediment Core Collection

Sediment sampling may be conducted with the use of self-contained core tubes, collected by SCUBA diver or ROV in accordance with the following methodology and quality criteria:

- Core samples will be collected using transparent core tubes, allowing a clear view of the entire sample surface
- Core tubes must be equipped with a means of either directly analyzing the top 2cm of sediment for required geochemical parameters or extracting a suitable subsample while maintaining an undisturbed sample surface
- Cores are to be inserted vertically, collecting a sample of at least 5 cm depth
- If the sample area is disturbed or contaminated in some manner, the dive crew shall select a new sample area as close as possible to the original station without sampling previously disturbed substrate
- Core samples are to be sealed as soon as possible following sample collection
- Cores must remain vertically oriented and maintain a relatively flat and undisturbed sediment-water interface until analysis is performed or a subsample is extracted
- Retrieved cores are to be photographed and evaluated for disturbance level (i.e. very clear with no disturbance, clear with minimal disturbance, cloudy with moderate disturbance or not clear and disturbed) after surfacing
- Any subsamples collected must contain a minimum of 5 mL of sediment and be clearly labeled with a station and sample ID
- If it is determined that an acceptable core sample cannot be collected within approximately 10 m of the sampling location due to sediment composition or consolidation, an alternate station is to be established at the cage on the same side of the array which contains the next highest biomass and which is not already the target of a sampling station
- If an alternate station can not be established which meets the above criteria, visual monitoring, as described in Section 3.2.2, will be conducted in lieu of sediment sampling at the initially selected location
- If it is determined that an acceptable core sample cannot be collected within approximately 10 m of the sampling location due to sediment composition or consolidation at the alternate location, the station will be considered a ‘hard bottom station’ for that sampling event. Visual monitoring, as described in Section 3.2.2, will be conducted in lieu of sediment collection

4.3 Sediment Storage and Transportation

Samples should be analyzed as quickly as possible following retrieval. If samples are to be stored and transported for analysis at a later time, the following guidelines must be followed:

- Sample storage containers used must not be made of any material or used in such a way that may negatively impact subsequent laboratory analysis
- Samples must be sealed against the intrusion of air and contain no apparent air bubbles throughout the sample
- A flexible, impermeable barrier, such as Parafilm® or Saran Wrap® should be used in addition to a tight fitting cap in order to ensure an air tight seal
- If headspace is unavoidable in a sample or subsample vessel, inert gas (e.g. nitrogen gas) may be used to cover the sample prior to closure of container

- As soon as possible following sample or subsample collection, sediments must be stored in the dark at 2-5ºC until they can be analyzed.
- A thermometer for immediate reference and a continuous temperature logger, recording at a minimum 30 minute interval, must accompany samples until laboratory analysis is conducted
- Sample temperature data must be maintained by the monitoring party and may be requested by NSDFA to assess quality assurance and quality control (QA/QC) of sediment storage and transport

4.4 Sediment Collection Observation Requirements

For each successful sediment retrieval attempt which meets the appropriate methodology and quality criteria outlined above, detailed observations are to be made, recorded and submitted to NSDFA. Observations recorded for each sample retrieval should include, but are not limited to:

- Sampling Observations:
    - Waterbody
    - Aquaculture lease number
    - Station ID and replicate number
    - Distance along established transect (where applicable)
    - Date and time of sample collection
    - Water depth
    - Latitude and longitude of sampling location
    - Type of sampling equipment utilized
    - Name of personnel collecting samples
    - Number of collection attempts required
    - Any deviations from prescribed standard operating procedures

- Sample Observations:
    - Sediment colour at surface and subsurface
    - Sample composition (e.g. mud, sand, cobble etc.)
    - Sample odor
    - Total sample depth
    - Macrofauna observed
    - Macroflora observed
    - Presence and relative abundance of uneaten finfish feed
    - Presence and relative abundance of finfish faeces
    - Presence and relative abundance of other organic detritus
    - Presence of gas bubbles released from sediment
    - Presence and relative abundance of *Beggiatoa* like bacterial mats
    - Presence and relative abundance of polychaete complexes
    - Presence and description of anthropogenic debris

5 ANALYSIS OF SEDIMENT SAMPLES

Information contained within this section provides guidance for the analysis of sediment samples for the Nova Scotia EMP. The procedures outlined below are based on information found in Wildish *et al*. (1999) and Wildish *et al*. (2004). Recent revisions have been made according to discussions and feedback from the April 2014 Nova Scotia Aquaculture Environmental Coordinating Committee (AECC) meetings.

The NSDFA has approved the Accumet AP63 and AP125 Portable pH/Ion Meter, Orion Silver/Sulfide Ionplus® Sure-Flow® Solid State Combination Ion Selective Electrode (Cat. No. 9616BNWP) and Orion Epoxy Sure-Flow Combination Redox/ORP Electrode (Cat. No. 9678BNW) for measurement of sulfide and redox. Once per year, prior to the initiation of EMP sediment analyses, the analytical party must submit to NSDFA, for approval, a list of chemicals (name and CAS#) and analytical equipment (name and model #) intended for EMP sediment analysis. Each instrument must be associated with a unique identifier and recorded. Laboratory records (e.g., logbooks, original records etc.) may be requested by NSDFA for QA/QC laboratory audits. A sample of the data recording sheet can be found in Appendix A4, respectively. Please retain original record of sampling data.

5.1 Redox Analysis (Eh)

Oxidation-reduction potential (redox), measured in millivolts (mV), is a measure of oxidation-reduction potential in sediments and is an indirect indicator of aerobic versus anaerobic conditions.

5.1.1 Materials

- Accumet AP63 or AP125 Portable pH/Ion Meter (Cat. No. 13-636-AP63 or 13-636-AP125)
- Orion Epoxy Sure-Flow Combination Redox/ORP Electrode (Cat. No. 9678BNW)
- Accumet ATC probe (Cat. No. 13-620-19)
- 4 M KCL saturated with Ag/AgCl (Cat. No. 900011)
- ORP standard (Cat. No. 967901 or 967961)
- Sampling receptacles (labelled, decontaminated and pre-weighed (g))
- Timer
- A3 data record sheet

5.1.2 ORP Electrode Accuracy Check

An accuracy check is to be performed **before and after** analysis using the commercially available ORP standard solution. The redox electrode must be filled with 4 M KCl saturated with Ag/AgCl at least 24 hours before use (Wildish et al., 1999). Place the electrode in a sample of 25°C ORP standard solution and record the mV reading. At 25°C, absolute mV values should equal 220 ± 3 mV. Accuracy check readings are to be recorded on the data recording sheet. Include notes regarding any errors or irregularities on data sheets. See Appendix A5 for a suggested procedure to detect coatings on the electrode platinum surface (this is not mandatory).

5.1.3 Redox Measurements

Triplicate subsamples taken from each monitoring station will be analyzed for redox in accordance with the protocol outlined below.

- Measurements will be completed within 72 hours of sample collection. If storage is required, samples must be stored in the dark, on ice (chilled, not frozen) in the field and transferred to a refrigerator held at 2 - 5°C (a temperature logger must be used to measure storage temperatures (see Section 4.0).
- From the cut-off 5 mL (or 10 mL) syringe, the first 2 mL (5 mL) are isolated from the upper 3 mL (5 mL) by first extruding 2 mL (5 mL) into a labelled, decontaminated, pre-weighed (g) receptacle for sediment porosity and percent organic matter analysis. The upper 3 mL (5 mL) are extruded into a separate labelled, decontaminated, pre-weighed (g) receptacle for redox and sulfide analysis.
- Receptacles used for redox and sulfide analysis should have a volume capacity that minimizes headspace.
- Measurements will be taken with Accumet AP63 or AP125 Portable pH/Ion Meter, Orion Epoxy Sure-Flow Redox/ORP Electrode and Accumet ATC probe.
- The redox probe should be held stationary during analysis. Hold the probe firmly in place below the sediment surface (Hargrave, personal communication).
- Redox measurements will be recorded as millivolts relative to the normal hydrogen electrode (mV_{NHE}) using the equation $mV_{NHE} = Eo + (224 - T)$, where Eo = mV of unknown and T = temperature of unknown (°C). Record the mV and temperature readings once the mV value has stabilized (stable reading displayed on meter or mV drift is < 10 mV/minute). If stabilization is not achieved, record the mV and temperature values when 2 minutes has elapsed (use a timer to achieve consistency among samples). Note on A3 data sheet which readings were taken at the 2 minute mark.
- The redox electrode will be rinsed with distilled water and dried between measurements (gently blot dry with Kimwipe).
- Redox and sulfide measurements must occur sequentially on one subsample before commencing redox analysis on the next subsample.
- All replicate 1's from each sampling location must be analyzed first, followed by all replicate 2's and then 3's to disperse evenly across all samples any potential influence that probe drift may have on measurements throughout the period of analysis.
- The order of subsample analysis, based on station ID, should be the same when each replicate group is analyzed.

5.2 Sulfide Analysis

Total dissolved sulfide, measured in micromolar (μM), is a measure of the accumulation of soluble sulfides, a major product of sulfate reduction that occurs under anaerobic conditions. This is a sensitive indicator of habitat degradation due to organic loading and currently the main indicator currently used to determine direct impact of an aquaculture operation.

As an accuracy check for the internal meter calculation, record the associated millivolt (mV) value for **both** the calibration and sulfide analysis. This allows calculation of sulfide concentrations directly from the calibration curve.

5.2.1 Materials

- Accumet AP63 or AP125 Portable pH/Ion Meter (Cat. No. 13-636-AP63 or 13-636-AP125)
- Orion Silver/Sulfide ionplus® Sure-Flow® Solid State Combination Ion Selective Electrode (Cat. No. 9616BNWP)
- Accumet ATC probe (Cat. No. 13-620-19)
- Orion Optimum Results B filling solution (Cat. No. 900062)
- Sodium sulfide (Na_2S) standards (100, 500, 1000, 5000, 10000 µM)
- Sulfide antioxidant buffer (SAOB) + L-ascorbic acid
- A3 data record sheet

5.2.2 Sulfide Electrode Calibration

Five sodium sulfide standards will be used to calibrate the sulfide electrode prior to sample analysis (100, 500, 1000, 5000 and 10000 µM). Sodium sulfide standards are unstable and oxidize readily in aerobic conditions and should be prepared fresh with deaerated water (distilled or deionized). SAOB + L-ascorbic acid are combined and added to standards just prior to calibrating. See Wildish et al. (1999) for preparation of sodium sulfide standards and SAOB + L-ascorbic acid solution. An exothermic reaction is initiated during the preparation of SAOB; therefore this solution must be cooled to 2 – 5 °C prior to use. See the electrode and meter manuals for calibration steps (Thermo Scientific, 2007b and Fisher Scientific, 2009).

The sulfide electrode will be filled with Orion Optimum Results B filling solution at least 24 hours before use (Wildish et al., 1999);

- SAOB is stable for a maximum of 3 hours following the addition of L-ascorbic acid (Wildish et al., 1999). If the SAOB + L-ascorbic acid solution exhibits a colour change prior to the 3 hour expiration, it is recommended to prepare a fresh solution. Record time that L-ascorbic acid is added to SAOB and time solution expires or colour change is observed on A3 data sheet.
- Always dilute standards using a 1:1 ratio with SAOB + L-ascorbic. Do not add SAOB + L-ascorbic acid to standards until just prior to calibration.
- Standards should not be shaken, rather gently swirled or stirred to adequately mix the SAOB + L-ascorbic acid and standard.
- Each standard and SAOB + L-ascorbic acid solution must reach the **same** target temperature (between 20-25°C) before calibrating the electrode.
- Follow the meter calibration steps (Fisher Scientific, 2009). Record both µM and mV readings once the target temperature is reached. Also, record the displayed slope value for the 10,000 µM standard on the A3 data sheet (the acceptable range -27 to -33 mV).
- The Accumet AP63 and AP125 Portable pH/Ion meter's default calibration values are a factor of 10 times less than the actual standard concentrations; therefore, the displayed calibration value must be multiplied by 10 to obtain the correct concentrations.
- Calculate the 10-fold mV change (slope). This value provides the best means for checking electrode operation (see Thermo Scientific, 2007b).
    - mV (5000 µM) – mV (500 µM) = 10-fold mV change.
    - mV (10000 µM) – mV (1000 µM) = 10-fold mV change.
    - The acceptable value range is -25 to -30 mV.

- Include notes regarding any calibration problems on A3 data sheet.
- Calibration of the sulfide electrode is stable for a maximum of three hours. Record time calibration completed and time of expiry on the A3 data sheet.

5.2.3 Sulfide Measurements

Triplicate subsamples taken from each monitoring station will be analyzed for sulfide in accordance with the protocol outlined below.

- Measurements will be completed within 72 hours of sample collection (Wildish et al., 1999).
- Measurements will be taken with Accumet AP63 or AP125 Portable pH/Ion Meter, Orion Silver/Sulfide ionplus® Sure-Flow® Solid State Combination Ion Selective Electrode and Accumet ATC probe.
- Receptacles used during analysis should have a volume capacity that minimizes headspace.
- Always dilute samples using a 1:1 ratio with SAOB + L-ascorbic. (i.e., each 3 mL sediment subsample will be mixed with 3 mL of SAOB + L-ascorbic acid).
- Samples should not be shaken, rather gently swirled or stirred to adequately mix the SAOB + L-ascorbic acid and sample.
- Sulfide readings will be taken once the SAOB + L-ascorbic acid and sample mixture reaches the same temperature at which the electrode was calibrated, and stabilization is achieved (‘stable’ displayed on meter). Note samples that are up to temperature but have not stabilized within 2 minutes. Record µM and mV values. Multiply µM values by a factor of 10 and record as ‘adjusted’.
- The sulfide electrode is to be rinsed with distilled water and dried between sample measurements (gently blot dry with Kimwipe).

5.3 Sediment Porosity

Porosity is the percentage (%) of pore volume or void space, or the volume within any material (e.g., bottom sediment) that can contain fluids. Porosity is an indirect measure of grain size and is used to detect changes in sediment consistency which may result from sedimentation of faeces and excess feed.

The method described below is to be performed using a gravity convection drying oven (e.g., Lindberg/Blue M 260) and an analytical balance (e.g., Denver Instrument Summit Series, SI 234); *other make/models are acceptable*:

5.3.1 Materials

- Gravity convection drying oven
- Analytical balance
- Labelled, pretreated, pre-weighed (g) receptacles
    - Glass receptacles must be acid washed between analyses to avoid cross contamination.
    - Receptacles used for both porosity and organic matter analysis must be pre-ashed before sediment is introduced.
- Vacuum desiccator
- Worksheet

5.3.2 Porosity Measurements

- Pre-heat drying oven to 60 °C.
- Record wet weight (g) of pre-weighed receptacle and sediment sample.
- Place weighed receptacles and sediment in the drying oven for 24 hours at 60 °C.
- Following 24 hours, place dried samples in a vacuum desiccator to bring to room temperature prior to weighing.
- Record dry weight (g) of receptacle and sediment sample. Weight recordings (g) should be recorded to at least 4 decimal places. The porosity value can be calculated as a percentage of the total volume of material:

(Wet sediment and receptacle weight) – (receptacle weight) = Wet sediment weight (g)
(Dry sediment and receptacle weight) – (receptacle weight) = Dry sediment weight (g)

[(Wet sediment weight – Dry sediment weight) / Wet sediment weight] x 100 = porosity (%)

5.4 Sediment Percent Organic Matter (POM)

Organic matter is observed to determine the portion (%) of sediment that is of plant or animal origin (combined). This variable is a good measure of organic loading.

The method described below is to be performed on the pre-dried samples from porosity analysis (section 6.3) using a muffle furnace (e.g., Barnstead/Thermolyne, Type 48000); *other make/models are acceptable*:

5.4.1 Materials

- Pre-ashed receptacles (labelled and pre-weighed (g))
- Tweezers
- Ceramic tray
- Muffle furnace
- Analytical balance
- Vacuum desiccator
- Worksheet

5.4.2 Percent Organic Matter Measurements

- Handling the labelled, pre-weighed (g), pre-ashed receptacle with tweezers, add approximately 0.5 g of ground, homogenized, dried sediment from the porosity analysis to the muffle furnace-safe receptacle. Record the weight. Weight recordings (g) should be recorded to at least 4 decimal places.
    - Sample homogenization is only required if the dried sediment is subsampled for POM measurements. Take care to avoid cross contamination between samples.
- Place samples in a cold muffle furnace. Set muffle furnace to 490 °C for 8 hours.
- Allow furnace to cool down before handling samples. Place ashed samples in a vacuum desiccator to bring to room temperature prior to weighing.
- Record weight of receptacle and ashed sediment sample.
- Percent organic matter can be calculated as follows:

Dried sediment – ashed weigh boat = Dried sediment (g)
Ashed sediment – ashed weigh boat = Ashed sediment (g)
Dried sediment – ashed sediment = Sediment organic content (g)

Sediment organic content (g) / Dried sediment (g)] x 100% = organic matter (%)

6 RECORD KEEPING

NSDFA will review all environmental monitoring performed as part of this program. Pre-sampling submissions are required to be submitted to NSDFA a minimum of two weeks prior to sampling. Data submissions are required to be submitted to NSDFA 14 and 21 days following sample collection. In summary, the final submission must include:

- **Pre-sampling**
    - Once a year: A list of chemicals (name and CAS#) and equipment (model name and #) intended for use for the EMP season.
    - 2 weeks prior to sampling: Electronic site diagram (kg fish/cage and location of proposed monitoring stations), proposed sampling methodology and tentative sampling date.
- **Within 14 days of sediment collection:**
    - A1 – Coordinate and Lab Results Table
        - All data fields completed except for porosity and organic matter
    - A3 – Analytical Data Record Sheet
        - Site name/#, date of sampling and analysis etc., redox probe accuracy check and sulfide calibration results (redox and sulfide sediment results will be included in A1)
- **Within 21 days of sediment collection:**
    - A1 – Coordinate and Lab Results Table (completed)
        - All data fields completed
    - A2 – Video and Sediment Sampler Log Sheet (1 per station)
    - Sediment sample photos (both sides of the grab shown)
    - Video recordings

7 BASELINE MONITORING

Baseline data collection is required in the following situations:

- The establishment of new site
- Reactivation of a lease
- Amendment of the boundaries of an existing, active lease

Collection of appropriate and complete baseline data ensures that ongoing environmental monitoring data can be comparted with the initial condition of the site. The following sections outline the information to be collected and methodologies used in order to comply with NSDFA requirements for Baseline Monitoring. Two hard copies and an electronic copy of the required information and video must be sent to the attention of the Manager of Aquaculture Operations at the following address:

Manager, Aquaculture Operations
Nova Scotia Department of Fisheries and Aquaculture, Aquaculture Division
1575 Lake Road
Shelburne, NS
B0T 1W0

7.1 Baseline Monitoring Requirements

7.1.1 Finfish Requirements

For a typical marine shellfish site, baseline monitoring will consist of the following:

- Video collection at each vertex of the proposed lease boundary
- Video collection at the center of the proposed lease boundary
- Video collection at a reference station located between 100 and 300 meters from the proposed lease boundary in the direction of the dominant current
- Sediment collection and analysis at the above monitoring stations
- Video collection along a transect running through the centre of the entire length of the proposed lease, or center of the proposed expansion area
- Collection of current speed and direction measurements for 30 days
- A bathymetry survey of the proposed lease area

Baseline information for marine finfish sites is also required by the Department of Fisheries and Oceans Canada (DFO) under the *Aquaculture Activity Regulations*. For more information, please visit their website or contact DFO's Aquaculture Management office. Sites which have been subject to a fish and fish habitat survey, as described in the *Aquaculture Activities Regulations*, may be exempt from the video collection requirements specified in this section.

Additional data collection may be required during baseline sampling at the discretion of NSDFA as a result of bay-specific risk-assessment. Proponents will be informed of any additional sampling requirements prior to the approval of a baseline sampling plan.

7.1.2 Shellfish Requirements

The majority of marine shellfish sites in Nova Scotia have historically posed minimal environmental risk and, therefore, warrant a reduced level of baseline monitoring. For a typical marine shellfish site, baseline monitoring will consist of the following:

- Video collection at each vertex of the proposed lease boundary
- Video collection at the center of the proposed lease boundary
- Video collection at a reference station located between 100 and 300 meters from the proposed lease boundary in the direction of the dominant current
- Video collection along a transect running through the centre of the entire length of the proposed lease, or centre of the proposed expansion area.

Additional data collection may be required during baseline sampling at the discretion of NSDFA as a result of bay-specific risk-assessment. Proponents will be informed of any additional sampling requirements prior to the approval of a baseline sampling plan.

7.2 Video Collection

Video collection that is carried out at static Baseline Monitoring stations (i.e. lease vertexes, center and references) is to be conducted in a manner which satisfies the methodology and quality criteria presented in Section 3.1.

Collection of benthic video for a required transect through a proposed lease may be conducted via surface deployed drop camera, handheld, diver operated video camera or remotely operated vehicle. Transect video is required to meet the methodology and quality criteria presented in Section 3.1 as well as the following:

- Recordings will be conducted along a transect extending through the center and running the entire length of the proposed lease.
- If utilizing a surface deployed drop camera, video stations will be established along the length of the transect, at 100 m intervals
- A 360º panorama is only required at the beginning of each individual recording (i.e. if the camera system is recording continuously throughout multiple video stations of a transect a panorama is only required prior to submersion at the first station.)
- If utilizing a diver-held or ROV mounted camera to conduct a continuous transect, a weighted drop line or other visual guide must be used to mark the transect line as well as the individual video stations at 100m increments.
- A diver or ROV conducting a continuous transect will do so at a speed and height above bottom which allows for the clear observation and identification of macro flora and fauna within 1 meter to either side of the established transect line.

For all video collected as part of Baseline Monitoring, detailed observations, as outlined in Section 3.3, must be recorded. High quality copies of the original, unedited footage as well as their associated observations are to be provided to DFO (Aquaculture Management Office) and NSDFA (Aquaculture Division).

7.3 Sediment Collection

Benthic sediment collection is required at the vertexes, center and reference station associated with any finfish aquaculture lease undergoing Baseline Monitoring and may be performed using surface deployed equipment (e.g. dredge, grab, gravity corer) or manually operated core tube (diver-held or ROV). Collection, storage and transport of these samples is required to meet all of the methodology and quality criteria presented in Section 4 which apply to the chosen sampling method.

For each successful sediment retrieval attempt which meets the appropriate methodology and quality criteria, detailed observations, as outlined in Section 4.4, must be recorded and submitted to NSDFA.

If sediment consolidation or composition has resulted in five failed collection attempts before three acceptable samples can be retrieved, the station will be considered a ‘hard bottom station’ for that sampling event.

7.4 Sediment Analysis

All sediment samples collected as part of Baseline Monitoring are required to undergo laboratory analysis for free sulphide concentration, redox potential, percent organic matter, porosity and sediment grain size. Sediment analysis is required to meet the methodology and quality guidelines presented throughout Section 5.

7.5 Current Speed and Direction

Where measurements of the current are required for Baseline Monitoring, a detailed profile of speed and direction must be collected at the centre of the proposed lease using an Acoustic Doppler Current Profiler (ADCP) of appropriate specification for the location. Profiles of the entire water column, in bins of no greater than 1 meter, are to be recorded at intervals of 30 minutes or less, for a minimum of 30 days. Speed and direction data from each profile will be composed of a sufficient number of individual measurements (pings), averaged over an appropriate interval such that the expected standard deviation of reported current measurements is <1 cm/s. The instrument to be used must be correctly calibrated as per manufacture’s specifications and the results submitted to NSDFA along with details of the unit’s configuration setup and all raw data resulting from the deployment.

7.6 Bathymetry Survey

A bathymetric survey must be conducted in order to generate contours of depth, relative to chart datum, with a minimum resolution of 10 m across the entire lease area. A bathymetric chart from the Canadian Hydrographic Service that includes depth profile contours in 10 m increments may be used instead of conducting a bathymetric survey.

APPENDIX A: ASSOCIATED FIELD AND ANALYTICAL SHEETS

The following appendices include templates and guidance documents to be used as part of the standard operating procedures.

Appendix A1 includes a coordinate table to record and submit all coordinates used to determine precise monitoring station locations. This template also includes columns to input summary laboratory results.

Appendix A2 is a monitoring equipment decision tree.

Appendix A3 is a log sheet to record field notes.

Appendix A4 is a log sheet to record information recorded by video transects at hard bottom stations

Appendix A5 is a data worksheet to record the redox accuracy check, sulfide calibration and measured values of redox potential and sulfide in sediment samples.

Appendix A6 is a suggested procedure for pre-season preparation and on-going use of ORP electrodes.

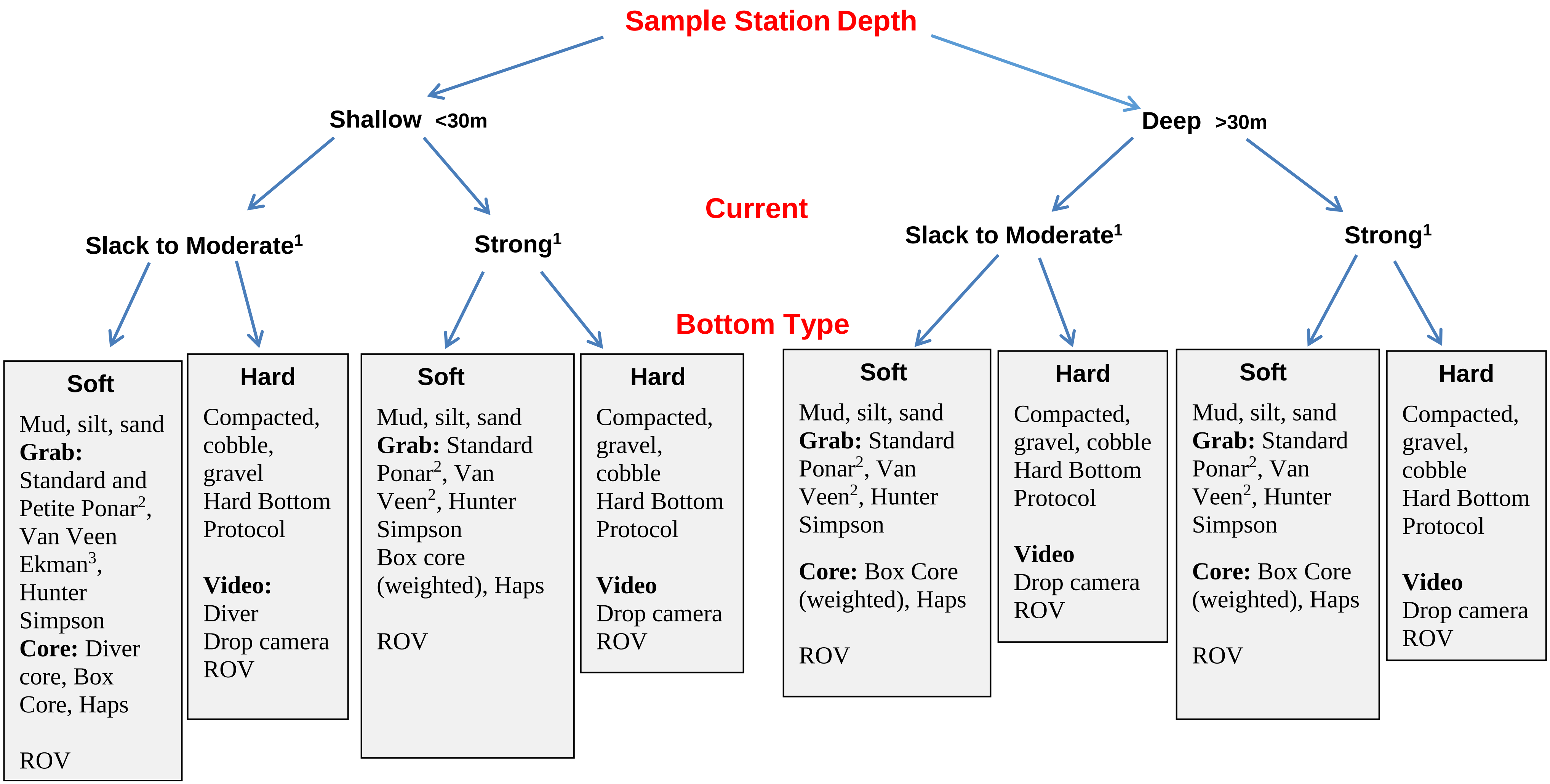
Appendix A7 is a checklist which outlines all information pieces required for submission.

APPENDIX A1: Coordinate and Lab Results Template

This template should be submitted in editable electronic spreadsheet format (i.e., excel) for all sampling events including baseline and Level I to III monitoring events. The coordinates should be submitted in NAD83 CSRS (decimal degrees). This template also includes columns to input summary laboratory results. Please submit this table with completed laboratory analysis of sample temperature, redox potential, total dissolved sulfide, porosity and percent organic matter. Data pertaining to individual replicates must be provided.

Monitoring Date	**Sample ID**		**Longitude**	**Latitude**	**Location**	**Lease #**	**Sample temp. (°C)**	**Redox (mV)**	**Redox (mV_{NHE})**	**Sulfide (µM) adjusted**	**Sulfide (mV)**	**Porosity (%)**	**Organic Matter (%)**
	Station ID	**Replicate**											
8-Aug-13	NSH01	1	43.33333	65.55555	Scotia Bay	0001							
8-Aug-13	NSH01	2	43.33333	65.55555	Scotia Bay	0001							
8-Aug-13	NSH01	3	43.33333	65.55555	Scotia Bay	0001							
8-Aug-13	NSH02	1	43.44444	65.66666	Scotia Bay	0001/Ref							
8-Aug-13	NSH02	2	43.44444	65.66666	Scotia Bay	0001/Ref							
8-Aug-13	NSH02	3	43.44444	65.66666	Scotia Bay	0001/Ref							

APPENDIX A2: Decision Guidelines for Selecting Monitoring Equipment



Sample Station Depth

Current

Bottom Type

Shallow <30m

Slack to Moderate[1]

Soft

Mud, silt, sand
Grab: Standard and Petite Ponar[2], Van Veen Ekman[3], Hunter Simpson
Core: Diver core, Box Core, Haps

ROV

Hard

Compacted, cobble, gravel
Hard Bottom Protocol

Video: Diver
Drop camera
ROV

Strong[1]

Soft

Mud, silt, sand
Grab: Standard Ponar[2], Van Veen[2], Hunter Simpson
Box core (weighted), Haps

ROV

Hard

Compacted, gravel, cobble
Hard Bottom Protocol

Video
Drop camera
ROV

Deep >30m

Slack to Moderate[1]

Soft

Mud, silt, sand
Grab: Standard Ponar[2], Van Veen[2], Hunter Simpson

Core: Box Core (weighted), Haps

ROV

Hard

Compacted, gravel, cobble
Hard Bottom Protocol

Video
Drop camera
ROV

Strong[1]

Soft

Mud, silt, sand
Grab: Standard Ponar[2], Van Veen[2], Hunter Simpson

Core: Box Core (weighted), Haps

ROV

Hard

Compacted, gravel, cobble
Hard Bottom Protocol

Video
Drop camera
ROV

[1] As a guide slack to moderate is considered to be 0-1 knot (0-0.5 m/s) while strong is greater than 1 knot (0.5m/s)..

[2] Thicker rubber flaps must be modified on the Ponar and Van Veen grab samplers so as to stiffen.

[3] Ekman not appropriate for use in moderate current.

APPENDIX A3: Video and Sediment Sampler Log Sheet

Date:			**Wind direction and speed:**	
Water body:			**Wave action:**	
Lease name and #:			**Direction and speed of current:**	
Monitoring Station ID:			**Tide schedule:**	
Latitude (decimal degrees):			**Video Notes:**	
Longitude (decimal degrees):				
Dist. and dir. from WP:				
Time:			**Comments:**	
Recorder name:				
Sample collector:				
Type of sediment sampler:			**Benthic Descriptor Key:**	
Station Depth (m):			1. Oxic layer thickness, gas bubbles, feed, faeces, sediment: colour, type and consistency	
Gear Present on Bottom (Description)			2. Degree of odour (strong, slight, none)	
Video (Y/N):			3. Flora/Fauna (e.g., eel grass, kelp, lobster, crab, starfish, *Beggiatoa*, polychaetes etc.)	
# sediment collection attempts:				

Sediment Samples	**Sample (y/n)**	**Sample ID**	**Sediment Sampler Retrieval Speed (cm/s)**	**Sediment Description[1]**	**Sediment Sample Depth (cm)**	**Odour[2]**	**Flora / Fauna[3]**
Benthic Replicate A							
Benthic Replicate B							
Benthic Replicate C							

Key Terms of Video and Sediment Sampler Log Sheet

Date – Date sample was collected.

Water body – Bay or Harbour name.

Lease name/# - Lease name/NSDFA lease number

Monitoring Station ID - Indicate the predetermined station identification code (e.g. SBH03)

Latitude –Monitoring station coordinate in decimal degrees (hddd.ddddd°)

Longitude – Monitoring station coordinate in decimal degrees (-hddd.ddddd°)

Dist. and dir. from WP – Indicate the distance (m) and direction from the intended waypoint.

Time – Time sample was collected.

Recorder Name - Name of person taking notes.

Sample Collector/Diver(s) Name – Name of person who collected the sample using surface deployed sampler or diver who collected the core.

Type of sediment sampler – Sediment sampler type (i.e., core tube or grab type).

Station Depth – Station water depth (m) at time of monitoring.

Gear Present on Bottom- Note any visible gear that is related to the aquaculture operation on the bottom (i.e. nets)

Video (Y/N) – Indicate if video was successfully collected. If no video collected, note the reason.

 # sediment collection attempts – State the number of sediment sampler deployments, in total, per replicate sampling.

Wind direction and speed – Describe the relative wind direction (e.g., N, SE, etc.) and relative speed (e.g., 10 knots).

Wave action – Describe the relative water conditions (e.g., flat, chop, swell, etc.).

Direction and speed of current – Describe the relative direction path (e.g., N-S, SW-NE, etc.) and relative speed (e.g., 10 knots) of the predominant current.

Tide schedule – State the times of high and low tide.

Video Notes – Sediment type, consistency and colour. Presence of biota (flora and fauna), presence of gas bubbles, presence of fish feed and/or faeces.

Comments – Include any notes pertaining to site changes, sampling difficulties, anchoring/mooring, differences between observed seafloor conditions and collected sediment sample, notes regarding sampling difficulties, weather issues, deviations from the SOP, etc.

Sample (Y/N) – Indicate if a replicate sample was collected.

Sample ID - List identification number listed on replicate core.

Sediment Description – Describe sediment characteristics of sediment sample. See Benthic Descriptor Key'.

Sediment Sample Depth – The measurement of the depth (cm) of the sediment within the sampler.

Odour – Indicate degree of odour from the sediment (strong, slight, none). See 'Benthic Descriptor Key'.

Flora/Fauna – Describe flora/fauna characteristics collected along with sediment sample. See 'Benthic Descriptor Key'.

APPENDIX A4: Video Monitoring Transect - Summary of Observations for Station

	0 M	**10M**	**20M**	**30M**	**40M**	**50M**
GPS coordinates NAD83 CSRS						
Sediment colour (brown, black, grey)						
Sediment consistency (mud; clay; rock; cobble; sand/silt)						
Sediment surface consolidation (firm packed; consolidated but easily disturbed; unconsolidated but very easily disturbed)						
Gas bubbles (none; rare; some; prevalent)						
Beggiatoa presence/absence and % coverage						
Opportunistic Polychaete worm Complexes (OPC) presence/absence						
Presence of feed (none; rare; some; prevalent)						
Presence of feces (none; rare; some; prevalent)						
Macro fauna/flora (none; relative abundance of polychaetes, molluscs; echinoderms and crustaceans; note which species are in relative abundance)						
Presence of gear on bottom						

APPENDIX A5: Analytical data record sheet

Lease # / name: ____________________________

Date samples collected: _________________ Time samples collected (x – x): _____________________

Date of analysis: _____________ Analysis start time: ____________ Analysis end time: ____________

Analysis performed by: ____

Redox Probe Accuracy Check (25°C)

ORP standard (start): _________ ORP standard (finish): _________

Sulfide Calibration

Temperature target (20-25°C): __________

Na_2S standard concentration (µM)	mV	Meter calculated slope (mV) (acceptable range -27 to -33 mV	Calculated 10-fold change (mV)	Verification check	
				µM	mV
100			1. 5,000 mV – 500 mV 2. 10,000 mV – 1,000 mV Acceptable range: -25 to -30 mV		
500					
1,000					
5,000			1.		
10,000			2.		

Time calibration completed: __________ Time L-ascorbic acid added to SAOB: __________
Time calibration expires: __________ Time SAOB + L-ascorbic acid expires: __________

Sample ID	Replicate #	Temp. (°C)	Redox (mV)	Sulfide (µM) unadjusted	Sulfide (mV)	NOTES:

APPENDIX A6: Suggested procedure for pre-season preparation and on-going use of ORP electrodes

1. Use only a refillable combination ORP electrode for Eh potential measurements in sediments. Gel-filled electrodes are not suitable.
2. Fill the electrode filling chamber with 4 M KCL saturated with Ag/AgCl and let stand for at least 24 hours prior to use.
3. Use the electrode to determine potential values of the ORP standard solution. Rinse (distilled water) and dry electrode on transfer between solutions and after use.
4. Place the electrode in aerated seawater and check readings every min for 5 min. Potentials should stabilize with minimum variability (±10mV) within 5 min. If potentials do not
5. stabilize repeat step 2. Rinse and dry the electrode.
6. The electrode is ready to use.
7. Record the Eh potential in aerated seawater at the beginning and end of each day of use and enter the values on sample data sheets.
8. Check the level of the reference 4 M KCL filling solution in the electrode daily. If it falls below the filling hole add more solution to bring the level up to the hole.
9. If the electrode is to be unused for an extended period of time empty the filling chamber, rinse with distilled, deionized water and store the electrode dry.

Comments

A brand new, accurately performing ORP electrode should have NHE-corrected Eh potentials in aerated seawater between 400 and 500 mV. Used probes generally have a lower range (300-400 mV). The raw potential on the meter before applying the NHE correction should be approximately +250 (±50) mV for new probes and +150 (±50) mV for used probes. The potential will be variable and differ between electrodes reflecting the absence of strong redox reactions in aerated seawater and differences in surface properties of the Pt tip of each electrode.

It is especially important to perform this procedure prior to using a new electrode in order to determine baseline potential values under oxic conditions. This check should also be applied routinely (at least daily) to determine if the electrode has been poisoned during use (Wildish et al., 2004). The Pt tip of an ORP electrode can be polished to remove oxic coatings. The electrode's response should be compared to the initial baseline value on a regular basis to ensure that the surface of the Pt tip has not been altered or damaged during use.

If Eh potentials fall below expected values in aerated seawater and polishing does not correct the electrode response to expected potentials the orifice between the filling solution and Pt tip may have become blocked with sediment. Wildish et al. (2004) described cleaning procedures to ensure that the orifice is open. If the orifice is not blocked the Pt tip has become damaged and the electrode should be replaced (Wildish et al., 2004).

APPENDIX A7: Checklist

Pre-monitoring

- Once/year: Submit to NSDFA a list of chemicals (name and CAS#) and equipment (model name and #) intended for use for the upcoming EMP season. **(*Note: due to the time to acquire/order materials, submit this list a minimum of 30 days prior to the commencement of analyses).***
- Two weeks prior to monitoring: Submit to NSDFA for review, an electronic site diagram with kg fish/cage, location of proposed monitoring stations displayed (Section 2.1), the proposed monitoring methodology and tentative monitoring date.

Monitoring

- Underwater video recordings of the seafloor at each station with GPS overlay
- 3 deployments of sediment samplers per station, 1 syringe subsample/sediment sampler (3 sediment subsamples per station)
- Photographs of each sediment sample
- A3 – Video and Sediment Sample log sheet completed (1 per station)

Sediment analysis

- Redox
- Sulfide
- Porosity
- Organic matter
- A1 – Coordinate and Lab Results Table completed (excel)
- A5 – Analytical Data Record Sheet completed

Submissions and timelines

- Within 14 days of sediment collection:
    - A1 – Coordinate and Lab Results Table
        - All data fields completed except for porosity and organic matter
    - A5 – Analytical Data Record Sheet
        - Site name/# date of sampling and analysis etc., redox probe accuracy check and sulfide calibration results (redox and sulfide sediment results will be included in A1)
- Within 21 days of sediment collection:
    - A1 – Coordinate and Lab Results Table (completed)
    - A3 – Video and Sediment Sample Log Sheet (1 per station)
    - A4- Video Transect Log Sheet (1 per station)
    - Sediment sample photos
    - Video recordings

For further information on timelines for monitoring events, submissions and necessary mitigation please refer to Section 5.0 Annual Schedules of the EMP Framework document (PNS, 2020).

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Journal of FISH BIOLOGY

fsbi An International Society for Fish Biology

Journal of Fish Biology (2014) **85**, 621–644
doi:10.1111/jfb.12447, available online at wileyonlinelibrary.com

To fast or feed: an alternative life history for anadromous brook trout *Salvelinus fontinalis* overwintering within a harbour

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(Received 4 February 2014, Accepted 8 May 2014)

The seasonal feeding pattern of sea-run brook trout *Salvelinus fontinalis* was studied from November to May 2010–2012 in Antigonish Harbour, Nova Scotia, Canada (45° 38′ N; 61° 55′ W). Sixty-three *S. fontinalis* (mean ± s.d. fork length = 330 ± 70 mm and mass = 536 ± 351 g) captured had fed predominantly on fishes (Fundulidae and Gasterosteidae). Percentage of empty stomachs was highest during autumn (18%) and winter (22%) and lowest in spring (7%). Stomach fullness increased from autumn to a maximum during winter, relating to near-zero body temperatures which may have effectively stopped gastric evacuation. Although feeding occurred during winter (December to March), consumption rates were calculated as negative values, and subsequently returned to positive values in spring (April to May). The over-winter life-history strategy of this sea-run *S. fontinalis* population appears to be a feeding marine migration in which fish continually increase body condition, representing an alternative to the more common overwintering strategy of starvation in fresh water until spring.

Key words: body condition; body temperature; consumption rates; marine migration; seasonal diet; stomach fullness.

INTRODUCTION

The marine migration of salmonids is largely driven by more favourable feeding conditions leading to rapid growth, enhanced fecundity and increased survival (Nikolsky, 1963; Harden Jones, 1968; Northcote, 1978, 1984). Most marine migrations commence in spring with a return to fresh water for autumn spawning, and overwintering for iteroparous species, with migrants often decreasing or ceasing feeding activity within fresh water (Klemetsen *et al.*, 2003; Quinn, 2005; Thorstad *et al.*, 2011). There are, however, documented accounts of Atlantic salmon *Salmo salar* L. 1758, brown trout *Salmo trutta* L. 1758 and Arctic char *Salvelinus alpinus* (L. 1758) populations residing in brackish to full-strength sea water during winter (Jonsson & Jonsson, 2002;

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Rikardsen *et al.*, 2006; Jensen & Rikardsen, 2008, 2012; Lacroix, 2013). It has also been observed that some anadromous brook trout *Salvelinus fontinalis* (Mitchill 1814) return to estuaries in autumn and remain there over winter (Bigelow & Welsh, 1925; Smith & Saunders, 1958; Gaudreault *et al.*, 1982; Curry *et al.*, 2002; Morinville & Rasmussen, 2006). This appears to be the case for an anadromous population of *S. fontinalis* overwintering in Antigonish Harbour, Nova Scotia, Canada.

Few studies have documented the marine diet of anadromous *S. fontinalis*, and most of them only provide a description of prey items found in stomachs, such as crustaceans (especially *Crangon* sp., *Gammarus* sp. and *Mysis* sp.), insects, mammals (deer mouse *Peromyscus* sp.) and polychaetes *Nereis* sp. Fishes consumed included capelin *Mallotus* sp., hake *Urophycis* sp., killifishes (Fundulidae), *S. fontinalis*, sand lance *Ammodytes* sp., sculpins (Scorpaeniformes), smelt *Osmerus* sp., sticklebacks (Gasterosteidae), rock gunnel *Pholis* sp. and whitefishes *Coregonus* spp. (White, 1940, 1942; Dutil & Power, 1980; Gaudreault *et al.*, 1982; O'Connell, 1982; Morinville & Rasmussen, 2006). Morinville & Rasmussen (2006) found crustaceans and fishes to be the dominant prey items for first and second year migrants, respectively, with *S. fontinalis* >250 mm fork length, L_F, showing piscivory.

Only two studies in the north-west Gulf of St Lawrence (48° N) documented estuarine winter feeding (Gaudreault *et al.*, 1982; Morinville & Rasmussen, 2006). There are apparently no diet studies of *S. fontinalis* populations overwintering in estuaries of the southern Gulf (45° N). To address the lack of detailed knowledge on marine migration and life-history strategies of northern anadromous salmonids, including information during winter (Huusko *et al.*, 2007), this study aimed to (1) describe the diet of *S. fontinalis* from autumn to spring and (2) determine if *S. fontinalis* are feeding under sea ice during winter. To do this, seasonal changes in diet and feeding intensity (% empty stomachs, stomach fullness, consumption rates and body condition) of *S. fontinalis* captured within Antigonish Harbour from November to May were examined. It was hypothesized that *S. fontinalis* diet would vary with the seasons, with piscivory occurring in migrants >250 mm L_F, with winter feeding focused on *Osmerus* sp. that are abundant from December to January. Feeding intensity during winter was expected to decrease with cold water temperatures, with percentage of empty stomachs being the highest, and stomach fullness and consumption rates being the lowest. Body condition was expected to be lowest in autumn and increasing until the end of the migration in late-spring.

MATERIALS AND METHODS

STUDY SITE

Antigonish Harbour, Nova Scotia, Canada (45° 38′ N; 61° 55′ W), is a semi-enclosed, Y-shaped, estuarine system with three rivers draining into the inner harbour, the South River in its southern end and the West and Wrights Rivers in its western extent (Fig. 1). The inner harbour is characterized by a shallow (water depth, Z, <2 m), silted, cordgrass *Spartina* sp. salt-marsh delta containing numerous inlets, coves and tidally influenced ponds. The mid-harbour has shallow inlets and multiple islands with a deep basin ($Z_{max} = 11$ m) and channels ($Z < 3$ m) fringed with blue mussels *Mytilus* sp and American oysters *Crassostrea* sp. The outer harbour is dominated by a barrier beach with two channels ($Z < 5$ m) connecting to St George Bay, Gulf of St Lawrence. The outer harbour contains multiple islands, shallow shoals blanketed by eel grass *Zostera* sp. (Thériault *et al.*, 2006) and a large salt-marsh pond. The harbour is known to

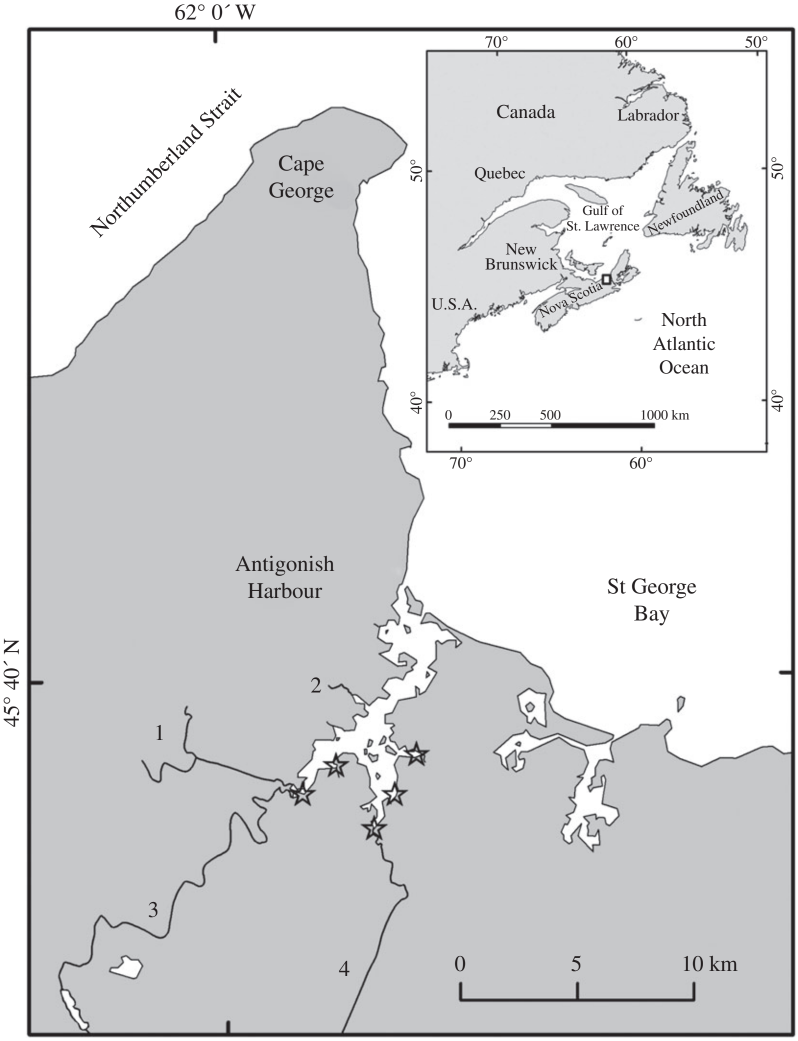

FIG. 1. Location of Antigonish Harbour, Nova Scotia, Canada, showing capture sites (☆) of sea-run *Salvelinus fontinalis* during late-autumn to mid-spring 2010–2012, and the (1) Wrights, (2) North, (3) West and (4) South Rivers.

contain a population of sea-run *S. fontinalis* which support a popular recreational spring fishery (MacMillan & Madden, 2007).

FISH SAMPLING

A total of 63 *S. fontinalis* were captured during day and night with a 5·1 cm stretched mesh gillnet, live-trap fyke net, rod and obtained opportunistically from fishers, from 19 November to 5 May 2010–2012. No captures occurred in February. Autumn samples were captured near the head of the tide in the South and West Rivers using fyke and gillnets ($n = 11$). Winter samples were captured by ice rod at the east harbour in January ($n = 6$) and by gillnet near the head of the tide in the West River in March ($n = 3$). Spring samples were caught throughout the inner harbour ($n = 43$), with the majority captured in West and South River estuaries (Fig. 1).

The L_F to the nearest mm and mass (M), including stomach contents, to the nearest 10 g were recorded. Excised stomachs were individually wrapped, labelled and stored on ice until freezing at −20° C. In the laboratory, stomachs were thawed, contents removed, separated and identified to family (Carlander, 1950; Borror & Delong, 1954; Needham & Needham, 1962; Bromley & Bleakney, 1984; Marshall, 2006) or alternative grouping (*i.e.* fish remains). Due to advanced digestion, some contents were numerically estimated from identifiable body parts (Power *et al.*, 2002). Excess moisture was blotted dry prior to counting and weighing (nearest 0·001 g).

STOMACH CONTENT ANALYSIS

Samples were pooled according to season: autumn (22 September to 20 December), winter (21 December to 19 March) and spring (20 March to 20 June), combining all collection years due to low sample sizes (Morinville & Rasmussen, 2006). For all samples and seasonal categories, indices of occurrence ($\%O$), number ($\%N$) and mass ($\%M$) were used to determine frequency and relative importance of each prey type consumed (Hyslop, 1980). The diet of an individual was quantified using the index of relative importance (I_{RI}) calculated for each taxon, excluding vegetal matter. The I_{RI} is defined as: $I_{RI} = (\%N + \%M)\ \%O$; where $\%N$ is the per cent number of certain prey item, $\%M$ is the per cent mass and $\%O$ is the frequency of occurrence (McLean *et al.*, 2013). The diets of *S. fontinalis* >250 mm L_F were examined for piscivory. Further seasonal diet differences were compared using total number of taxa, mean number of taxa and the Shannon–Weiner and Schoener overlap indices. The Shannon–Weiner diversity index (H') was calculated as: $\left(H'\right) = -\sum_{i}^{n} p_i \ln p_i$, where p_i is the mean $\%M \times 0{\cdot}01$ of taxon i. Diet overlap between seasons was compared with Schoener's overlap index (I_S), so that: $I_S = 1 - 0 \cdot 5 \left(\sum_{i=1}^{n} \left| p_{xi} - p_{yi} \right| \right)$, where p_{xi} is the mean $\%M$ of prey taxon i for season x and p_{yi} is the mean $\%M$ of prey taxon i for season y. The index varies from 0 (no overlap) to 1 (complete overlap; Guilbard *et al.*, 2007). Any fish with an empty stomach was excluded from stomach content analyses.

BODY TEMPERATURE (T_B)

Mean seasonal and interval body temperatures (T_B) of *S. fontinalis* were obtained from a concurrent tracking study using temperature–pressure sensor acoustic transmitters (V9TP and V13TP, Vemco/Amirix Ltd; www.vemco.com) and active (VR100) and passive receivers (VR2W and VR3-UWM), following the procedures outlined in the study of Spares *et al.* (2012). Simultaneous detections on multiple receivers and temperatures recorded beyond the transmitter's calibrated range and below saltwater freezing point (−1·9° C; DeVries & Cheng, 2005) were omitted.

SEASONAL FEEDING INTENSITY

To estimate feeding activity between seasons, percentage of empty stomachs (%ES) and indices for stomach fullness (I_R) and body condition factor (K) were calculated using all sampled fish (Rikardsen *et al.*, 2006), whereas means for number of prey, mass of prey (M_P), mass of vegetal and miscellaneous matter and consumption rate (C_{24}; Richter *et al.*, 2004) were calculated from fish with prey or vegetal and miscellaneous matter present in their stomachs (Guilbard *et al.*, 2007). The degree of stomach fullness was calculated using Hureau's index (I_R), where $I_R = 1000\ M_P\ M^{-1}$ [mg g^{-1}; Tudela & Palomera (1995); McLean *et al.* (2013)]. Food consumption rate of *S. fontinalis* was estimated as the daily ration (C_{24}; mg g^{-1}) using the Eggers method, so that $C_t = I_R R t + (I_{RT} - I_{RO})$, where C_t is the consumption over the feeding interval considered, I_R is the mean stomach fullness index for the interval, R is the instantaneous gastric evacuation rate (h^{-1}), t is the interval duration (h), I_{RO} is the mean stomach fullness index at the beginning of the interval and I_{RT} is the mean stomach fullness index at the end of the interval (Amundsen & Klemetsen, 1988; Tudela & Palomera, 1995; Rikardsen *et al.*, 2006). Six feeding intervals were considered: 19 November to 13 December, 13 December to 7 January, 7 January to 29 January, 29 January to 4 March, 4 March to 8 April and 8 April to 5 May. Mean I_{RO} and I_{RT} values were calculated from samples captured within 24 h of the beginning and end dates. The rate of gastric evacuation of *S. fontinalis* and its relation to temperature was estimated by Sweka *et al.* (2004) in laboratory studies using fly and beetle larvae as prey, and these estimates were adopted for this study. Fulton's condition factor (K) was calculated using $K = 10^5\ M\ L_F{}^{-3}$ (Ricker, 1975). A low K value (0·80) represented a fish in poor condition with a large head and narrow, thin body, whereas a large K value (1·60) indicated a fish in excellent condition with a well-proportioned, thick body, usually trophy class (Barnham & Baxter, 2003).

DATA ANALYSIS

Due to low sample sizes, a cumulative prey taxa abundance curve was generated by randomly ordering stomach samples to estimate the number of samples needed to compare taxa diversity between seasons. For all seasonal comparisons, either a parametric (ANOVA *post hoc* Tukey) or a non-parametric [Kruskal–Wallis with pair-wise comparisons using a Dwass-Steel-Chritchlow-Fligner (D-S-C-F)] test was conducted, with normal and non-normal sample distribution (Shapiro–Wilk test) determining the test, respectively (Baran & Warry, 2008). Outliers were identified using box-and-whisker plots generated by Systat 13 software and removed for hypotheses testing (Systat; www.systat.com). Means for %*O*, %*N* and %*M* were generated. An α level of 0·05 was used for all tests.

RESULTS

FISH SAMPLING

Mean ± S.D. L_F for all *S. fontinalis* sampled was 330 ± 70 mm ($n = 63$; Fig. 2). Mean ± S.D. M was 462 ± 282 g ($n = 56$), with fish masses unmeasured in the field removed from any statistical analysis involving M. Minimum and maximum L_F and M of *S. fontinalis* captured in autumn and spring were 179 and 450 mm, and 42 and 1235 g (Table I), respectively. There was a significant increase in L_F and M from autumn to spring (L_F, ANOVA *post hoc* Tukey, $F = 4{\cdot}93$, d.f. = 2, $P < 0{\cdot}01$, $n = 63$; $\sqrt{M}$, $F = 7{\cdot}545$, d.f. = 2, $P < 0{\cdot}001$, $n = 56$; Fig. 3).

STOMACH CONTENT ANALYSIS

Stomach content diversity expressed as total n of taxa present was highest in spring ($n = 32$) with the median number of taxa highest in autumn [2 (3·3)] and lowest in winter [1 (0·8); Table I; Kruskal–Wallis *post hoc* D-S-C-F: $t = 6{\cdot}76$, d.f. = 2, $P < 0{\cdot}05$, five outliers removed]. The cumulative prey taxa abundance curve revealed the maximum number of taxa to be 34 for the entire sampling period, and minimum of five taxa during winter, reaching these sample size asymptotes within 52 and three stomachs, respectively (Fig. 4). Stomach content indices revealed fish as the dominant component, occurring in 90·6% of all stomachs containing prey, as 58·7% by number of items, 93·1% wet mass of contents and >85 %*O* and >75 %*M* throughout all seasons, with the highest %*N* (76·1%) occurring in spring. Fundulidae [mummichog *Fundulus heteroclitus* (L. 1766)], Atherinopsidae [Atlantic silverside *Menidia menidia* (L. 1766)] and Gasterosteidae [*Apeltes quadracus* (Mitchill 1815), *Gasterosteus aculeatus* L. 1758,

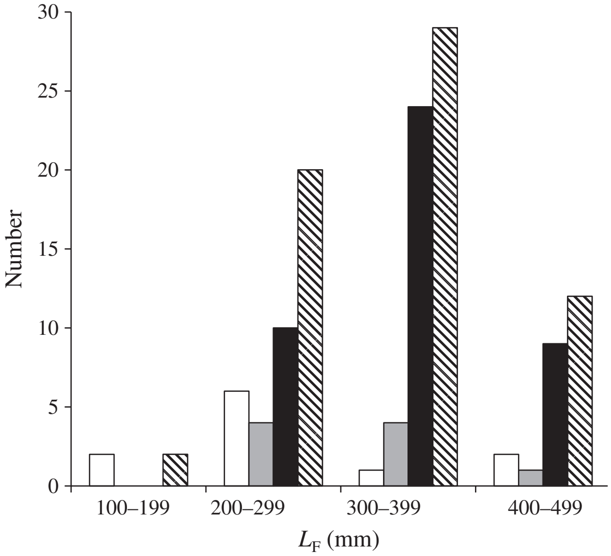

FIG. 2. Fork length (L_F) frequency of sea-run *Salvelinus fontinalis* captured during autumn ($n = 11$; □), winter ($n = 9$; ▨) and spring ($n = 43$; ■), and all data pooled ($n = 63$; ▧), from 2010 to 2012 in Antigonish Harbour, Nova Scotia, Canada.

Pungitius pungitius (L. 1758)] were consumed the most, constituting 14·4, 2·0 and 12·1 %*N*, and 69·3, 11·2 and 6·2 %*M*, respectively. Atherinopsidae only occurred in stomachs during winter and spring, constituting 10·0 %*N* and 22·9 %*M* of the winter diet (Table II and Fig. 5). Surprisingly, the largest prey item was an American eel *Anguilla rostrata* (LeSueur 1817) (220 mm total length, L_T, 22·8 g) coiled inside the stomach of a 448 mm L_F, 1235 g *S. fontinalis*. Seasonal specialities were Gammaridae in autumn (71·4 %*O*, 84·5 %*N* and 20·0 %*M*), Nereidae during winter (14·3 %*O*, 45·0 %*N* and 11·9 %*M*) and fish eggs in spring (2·6 %*O*, 42·1 %*N* and 0·2 %*M*; Fig. 5). Insects occurred during all seasons, and included 20 identified families, with caddis (Limnephilidae) and damsel (Coenagrionidae) flies, and ground (Carabidae) and scarab (Scarabaeidae) beetles consumed the most often (Table II).

For *S. fontinalis* > 250 mm L_F which contained prey ($n = 49$, 9% ES), piscivory accounted for 93·9 %*O*, 43·2 %*N*, 94·3 %*M* and 129·1% I_{RI}. Of the *S. fontinalis* < 250 mm L_F containing prey ($n = 7$, 22% ES), only one individual had consumed Fundulidae (14·3 %*O*, 0·3 %*N*, 18·5 %*M* and 2·7% I_{RI}; Fig. 6). Stomach contents of these smaller *S. fontinalis* included estuarine (Crangonidae, Gammaridae and Nereidae) and freshwater (Carabidae, Corydalidae, Dytiscidae, Limnephilidae and Lumbricidae) prey. One individual (213 mm L_F) had consumed 298 unidentified fish eggs (0·7 g wet mass), but these were not considered as fish for this analysis.

Percentage I_{RI} revealed Fundulidae (45·8%), Gasterosteidae (4·8%), Gammaridae (4·0%) and Atherinopsidae (2·2%) as the most valuable prey items year-round, with Gammaridae most prevalent in autumn (74·6%) and Fundulidae in winter (41·3%) and spring (53·9%; Fig. 5). The Shannon–Weiner diversity index was highest in autumn (1·28) and lowest in spring (1·06) with complete overlap (1·00) between seasons indicated by Schoener's overlap index (Table I).

TABLE I. Seasonal comparison of fork length (L_F), body mass (M), diet diversity, body temperature (T_B), feeding intensity and body condition (K) indices for sea-run *Salvelinus fontinalis* captured in Antigonish Harbour, Nova Scotia, Canada, during 2010–2012. Outliers were removed for all calculations, minimum and maximum values are given in parentheses

	Autumn	Winter	Spring	All
Mean ± s.d. L_F (mm)	278 ± 89 (179, 443)	317 ± 48 (256, 410)	346 ± 62 (213, 450)	330 ± 70 (179, 450)
n	11	9	43	63
Mean ± s.d. M (g)	270 ± 303 (42, 965)	363 ± 189 (140, 740)	540 ± 262 (150, 1235)	462 ± 282 (42, 1235)
n	11	8	37	56
Total *n* taxa	13	5	32	34
Median *n* taxa	2, 3·3 (1, 6)	1, 0·8 (1, 3)	1, 1 (1, 15)	1, 1 (1, 15)
n	9	7	40	56
H'	1·28	1·12	1·06	1·17
I_S		1·00	1·00	
Mean ± s.d. T_B (° C)	8·6 ± 3·6 (1·3, 20·1)	0·7 ± 1·2 (−0·5, 4·7)	9·0 ± 3·4 (0·2, 19·3)	7·6 ± 4·4 (−0·5, 20·1)
n	1147	9548	45 547	56 242
n with prey	9	7	40	56
n empty	2	2	3	7
% Empty stomachs	18	22	7	11
Median *n* prey	4, 4·5 (1, 10)	5, 4 (2, 18)	5, 9·8 (1, 28)	5, 8 (1, 28)
n	7	6	37	50
Median total prey mass (g)	0·44, 2·49 (0·001, 3·94)	14·00, 9·16 (1·25, 17·21)	5·37, 11·77 (0·05, 24·69)	4·78, 12·39 (0·001, 24·69)
n	8	7	40	55
n with vegetal and miscellaneous matter	3	3	8	14
Median mass of vegetal and miscellaneous matter (g)	0·09, 0·10 (0·05, 0·14)	0·30, 0·30 (0·11, 0·50)	0·09, 0·11 (0·04, 0·38)	0·11, 0·17 (0·04, 0·50)
n	2	3	7	12
Median I_R (mg g^{-1})*	0·6, 3·5 (0, 17·5)	22·2, 43·0 (0, 60·8)	12·7, 16·4 (0, 43·8)	11·5, 18·6 (0, 68·2)
n	9	8	36	53
Mean ± s.d. K	0·90 ± 0·12 (0·73, 1·11)	1·10 ± 0·16 (0·82, 1·25)	1·31 ± 0·15 (0·98, 1·61)	1·20 ± 0·22 (0·73, 1·61)
n	11	8	37	56

H', Shannon–Weiner index; I_S, Schoener's overlap index; I_R, stomach fullness.
*Includes empty stomachs.

BODY TEMPERATURE (T_B)

There was a significant difference between mean ± s.d. body temperatures of *S. fontinalis* for each season (ANOVA *post hoc* Tukey, $F = 27\,445$, d.f. = 2, $P < 0{\cdot}000$), calculated as 8·6 ± 3·6° C ($n = 1147$) during autumn, 0·7 ± 1·2° C ($n = 9548$) in winter and 9·0 ± 3·4° C ($n = 45\,547$) in spring, with an overall T_B of 7·6 ± 4·4° C (minimum = −0·5° C, maximum = 20·1° C, $n = 56\,242$; Table I).

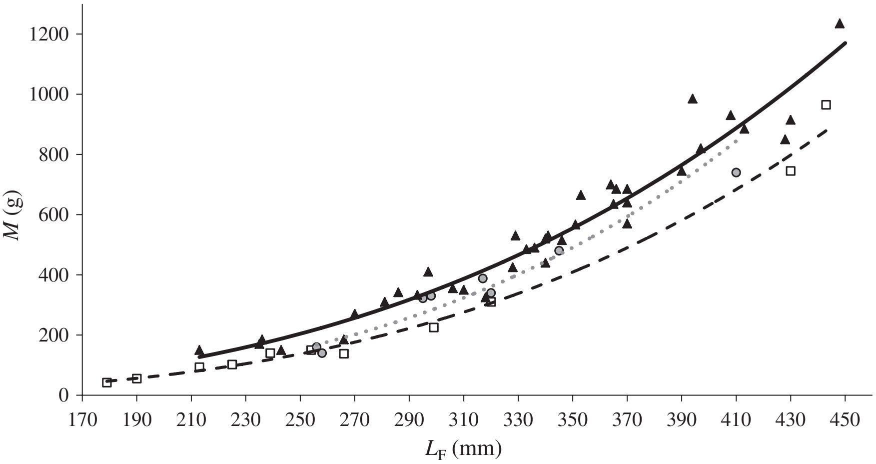

FIG. 3. Mass (*M*) and fork length (L_F) relationships for sea-run *Salvelinus fontinalis* ($n = 63$) captured from 2010 to 2012 during autumn (19 November to 13 December; □, – – –; $y = 2\text{E-}06x^{3{\cdot}24}$; $r^2 = 0{\cdot}988$, $n = 11$), winter (7 January to 5 March; ●, · · · ·; $y = 9\text{E-}07x^{3{\cdot}43}$; $r^2 = 0{\cdot}938$, $n = 9$) and spring (5 April to 5 May; ▲, ——; $y = 2\text{E-}05x^{2{\cdot}97}$; $r^2 = 0{\cdot}954$, $n = 43$).

FEEDING INTENSITY

Prey occurred in 56 of all *S. fontinalis* stomachs sampled: nine in autumn, seven in winter and 40 in spring. Percentage of empty stomachs was high during autumn (18%) and winter (22%) and lowest in spring (7%). The median number of prey per stomach did not change significantly with the seasons [four to five (4·0–9·8), Kruskal–Wallis test, $t = 0{\cdot}868$, d.f. = 2, $P > 0{\cdot}05$, $n = 50$, six outliers removed], whereas median M_P

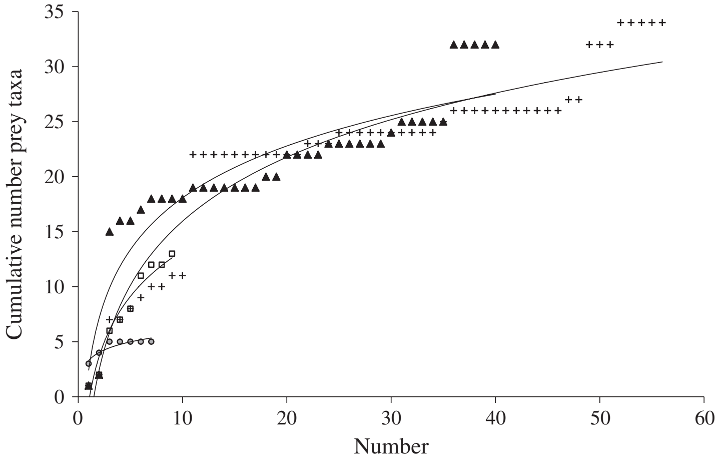

FIG. 4. Cumulative number of prey taxa for sea-run *Salvelinus fontinalis* containing prey during 2010 to 2012 ($n = 56$; +), autumn (19 November to 13 December; $n = 9$; □), winter (7 January to 5 March; $n = 7$; ●) and spring (5 April to 5 May; $n = 40$; ▲) with trend line asymptotes showing minimum sample size needed to reach maximum number of taxa.

TABLE II. Summary of stomach contents of *Salvelinus fontinalis* ($n = 63$) captured from November to May 2010–2012, in Antigonish Harbour, Nova Scotia, Canada. Indices used are frequency of occurrence (%*O*), per cent by number (%*N*) and per cent by mass (%*M*). Frequency of occurrence is expressed relative to the total number of stomachs containing food ($n = 53$)

	%*O*				%*N*				%*M*			
	All	Autumn	Winter	Spring	All	Autumn	Winter	Spring	All	Autumn	Winter	Spring
n	53	7	7	39	63	11	9	43	63	11	9	43
Estuarine prey items												
Actinopterygii	90·6	85·7	100·0	89·7	58·7	8·8	53·8	76·1	93·1	75·3	86·9	95·6
Anguillidae (freshwater eels)	1·9	0·0	0·0	2·6	0·1	0·0	0·0	0·1	5·2	0·0	0·0	6·7
Atherinopsidae (silversides)	17·0	0·0	14·3	20·5	2·0	0·0	10·0	1·8	11·2	0·0	22·9	9·3
Fundulidae (killifishes)	54·7	28·6	57·1	59·0	14·4	3·4	16·3	18·0	69·3	51·4	56·0	73·4
Gasterosteidae (sticklebacks)	26·4	28·6	42·9	23·1	12·1	3·8	27·5	13·2	6·2	21·5	8·0	4·8
Unidentified fish eggs	1·9	0·0	0·0	2·6	29·1	0·0	0·0	42·1	0·2	0·0	0·0	0·2
Unidentified fish remains	17·0	28·6	0·0	17·9	1·3	1·7	0·0	1·3	1·1	2·4	0·0	1·2
Crangonidae (sand shrimps)	9·4	0·0	0·0	12·8	1·0	0·0	0·0	1·4	1·0	0·0	0·0	1·3
Gammaridae (amphipods)	17·0	71·4	0·0	10·3	22·7	84·5	0·0	4·5	1·1	20·0	0·0	0·2
Nereidae (sandworms)	5·7	0·0	14·3	5·1	4·5	0·0	45·0	1·4	3·0	0·0	11·9	1·3
Freshwater prey items												
Amnicolida (aquatic snails)	1·9	14·3	0·0	0·0	0·2	0·8	0·0	0·0	0·0	0·1	0·0	0·0
Lumbricidae (earthworms)	1·9	0·0	0·0	2·6	0·2	0·0	0·0	0·3	0·3	0·0	0·0	0·4
Insecta (insects)	18·9	71·4	14·3	10·3	12·7	5·9	1·3	16·3	1·0	3·7	0·1	1·0
Coenagrionidae (damselflies)	5·7	14·3	0·0	5·1	1·9	0·4	0·0	2·5	0·1	0·3	0·0	0·2
Cicadellidae (leafhoppers)	3·8	0·0	0·0	5·1	0·3	0·0	0·0	0·4	0·0	0·0	0·0	0·0
Formicidae (flying ants)	3·8	0·0	0·0	5·1	0·3	0·0	0·0	0·4	0·0	0·0	0·0	0·0
Apidae (honeybees)	3·8	0·0	0·0	5·1	0·1	0·0	0·0	0·1	0·0	0·0	0·0	0·0
Coleoptera (beetles)	9·4	42·9	0·0	5·1	8·7	2·5	0·0	11·7	0·6	1·6	0·0	0·7
Carabidae (ground)	5·7	14·3	0·0	5·1	3·1	0·4	0·0	4·4	0·2	0·0	0·0	0·3
Dytiscidae (diving)	1·9	14·3	0·0	0·0	0·3	1·3	0·0	0·0	0·1	1·1	0·0	0·0
Staphylinidae (rove)	3·8	0·0	0·0	5·1	3·0	0·0	0·0	4·4	0·2	0·0	0·0	0·2

TABLE II. Continued

	%*O*				%*N*				%*M*			
	All	Autumn	Winter	Spring	All	Autumn	Winter	Spring	All	Autumn	Winter	Spring
Elateridae (click)	3·8	0·0	0·0	5·1	0·5	0·0	0·0	0·7	0·1	0·0	0·0	0·1
Scarabaeidae (scarab)	5·7	14·3	0·0	5·1	0·6	0·4	0·0	0·7	0·0	0·2	0·0	0·0
Hydrophilidae (water)	3·8	14·3	0·0	2·6	0·2	0·4	0·0	0·1	0·0	0·3	0·0	0·0
Curculionidae (weevils)	3·8	0·0	0·0	5·1	0·2	0·0	0·0	0·3	0·0	0·0	0·0	0·1
Chrysomelidae (leaf)	1·9	0·0	0·0	2·6	0·1	0·0	0·0	0·1	0·0	0·0	0·0	0·0
Coccinellidae (ladybugs)	1·9	0·0	0·0	2·6	0·1	0·0	0·0	0·1	0·0	0·0	0·0	0·0
Pentatomidae (stinkbugs)	1·9	0·0	0·0	2·6	0·1	0·0	0·0	0·1	0·0	0·0	0·0	0·0
Cydnidae (burrower bugs)	3·8	0·0	0·0	5·1	0·6	0·0	0·0	0·8	0·0	0·0	0·0	0·0
Corydalidae (fishflies)	1·9	14·3	0·0	0·0	0·2	0·8	0·0	0·0	0·0	0·8	0·0	0·0
Calliphoridae (blowflies)	1·9	0·0	0·0	2·6	0·1	0·0	0·0	0·1	0·0	0·0	0·0	0·0
Limnephilidae (caddisflies)	5·7	28·6	14·3	0·0	0·3	0·8	1·3	0·0	0·0	0·6	0·1	0·0
Leptophlebiidae (mayflies)	1·9	0·0	0·0	2·6	0·3	0·0	0·0	0·4	0·0	0·0	0·0	0·0
Acrididae (grasshoppers)	1·9	0·0	0·0	2·6	0·1	0·0	0·0	0·1	0·0	0·0	0·0	0·0
Unidentified insect remains	7·5	42·9	0·0	2·6	0·4	1·3	0·0	0·1	0·0	0·3	0·0	0·0
Araneae (spiders)	1·9	0·0	0·0	2·6	0·1	0·0	0·0	0·1	0·0	0·0	0·0	0·0
Unidentified/miscellaneous	26·4	42·9	42·9	20·5	–	–	–	–	0·9	7·0	1·2	0·5

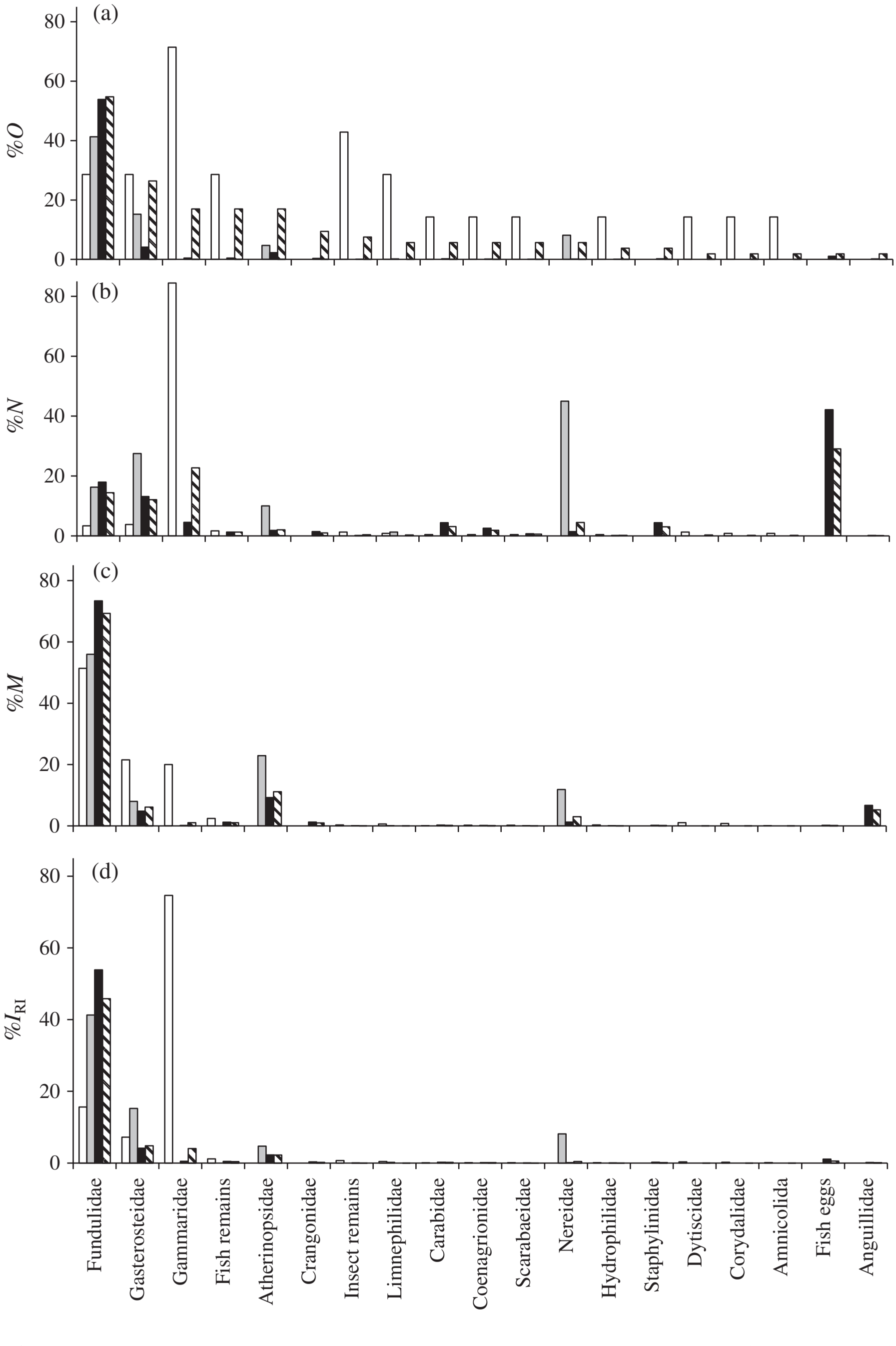

FIG. 5. (a) Relative per cent occurrence (%*O*), (b) abundance (%*N*) and (c) mass (%*M*) of taxa collected from the stomach contents of sea-run *Salvelinus fontinalis* ($n = 56$) captured during autumn (□), winter (▨) and spring (■), and all data pooled (▧), from 2010 to 2012 in Antigonish Harbour, Nova Scotia, Canada. (d) The per cent index of relative importance (% I_{RI}) of each taxa to *S. fontinalis* diet is also presented. Prey taxa with %$O < 10$ were not included but the following order (Araneae) and families: Acrididae, Apidae, Calliphoridae, Chrysomelidae, Cicadellidae, Coccinellidae, Curculionidae, Cydnidae, Elateridae, Formicidae, Leptophlebiidae, Lumbricidae and Pentatomidae were recovered.

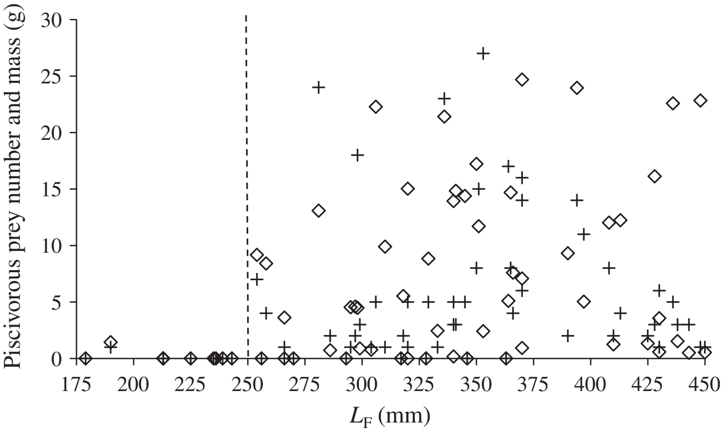

FIG. 6. Fork length (L_F) and piscivory shown by number (+) and wet mass (◇) of fishes consumed by sea-run *Salvelinus fontinalis* captured in Antigonish Harbour, Nova Scotia, Canada, from November to May 2010–2012.

was significantly higher in winter [14·00 g (9·16), $n = 7$], compared to spring [5·37 (11·77), $n = 40$], and autumn [0·44 (2·49), $n = 8$, one outlier removed; Kruskal–Wallis *post hoc* D-S-C-F, $t = 11·76$, d.f. = 2, $P < 0·05$]. The percentage of stomachs containing vegetal and miscellaneous matter was highest in winter (43%) and lowest in spring (20%) with no significant seasonal difference in median vegetal mass (Kruskal–Wallis test, $t = 2·89$, d.f. = 2, $P > 0·05$, two outliers removed; Table I).

There were significant seasonal differences (Kruskal–Wallis *post hoc* D-S-C-F, $t = 8·30$, d.f. = 2, $P \leq 0·02$) in I_R with the lowest median index of 0·6 in autumn [(interquartile range = 3·5), $n = 9$, two outliers removed], 12·7 during spring [(16·4), $n = 36$, one outlier removed] and highest of 22·2 in winter [(43·0), $n = 8$; Table I]. Further divisions into approximately monthly intervals (28 ± 5 days) revealed mean stomach fullness indices following a similar pattern (Fig. 7), beginning at a mean ± S.D. of 11·6 ± 21·3 from 19 November to 13 December, increasing to a maximum of 32·6 ± 24·7 from 29 January to 4 March and decreasing to 11·7 ± 11·5 from 4 March to 8 April (Table III). Estimates of consumption rate (C_{24}) revealed a reciprocal relationship (Fig. 7) with the highest values of 3·67 and 5·93 mg g^{-1} occurring during 19 November to 13 December and 8 April to 5 May, respectively, and negative consumption values occurring from 7 January to the 4 March (Fig. 7). Negative consumption corresponded to recorded body temperatures of −0·5 to 3·8° C (Table III).

Mean ± S.D. K of *S. fontinalis* increased significantly from 0·90 ± 0·12 in autumn ($n = 11$) to 1·10 ± 0·16 in winter ($n = 8$), and peaked at 1·31 ± 0·15 ($n = 37$) in spring (Table I and Fig. 8; $r^2 = 0·577$, $n = 56$; ANOVA *post hoc* Tukey, $F = 34·63$, d.f. = 2, $P < 0·05$). Minimum and maximum K indices of 0·73 and 1·61 occurred during autumn and spring, respectively (Table I).

DISCUSSION

Salmonid populations feeding in marine environments have demonstrated differences in energy allocation to somatic growth, body maintenance, gonad development and

TABLE III. Summary of calculations (mean ± S.D.; minimum and maximum in parentheses) used to estimate consumption rates (C_{24}) of sea-run *Salvelinus fontinalis* during seasonal intervals from 19 November to 5 May 2010–2012, in Antigonish Harbour, Nova Scotia, Canada. Calculations include body temperature (T_B), stomach fullness at beginning of interval (I_{RO}), stomach fullness at end of interval (I_{RT}), stomach fullness over entire interval (I_R), instantaneous gastric evacuation rate [R, estimated from Sweka *et al.* (2004)] and interval duration (t)

Interval	19 November to 13 December	13 December to 7 January	7 January to 29 January	29 January to 4 March	4 March to 8 April	8 April to 5 May
T_B (° C)	6·3 ± 1·7	3·9 ± 1·6	1·4 ± 1·3	0·1 ± 0·2	3·5 ± 2·6	7·4 ± 2·5
	(3·0, 9·1)	(0·5, 9·4)	(0·02, 3·8)	(−0·5, 2·1)	(−0·1, 11·5)	(0·7, 14·9)
n	413	754	2208	5564	7158	23 108
I_{RO} (mg g^{-1})	1·22 ± 1·71	18·3 ± 29·0	15·0 ± 21·2	52·5 ± 11·8	19·3 ± 22·1	10·3 ± 6·4
	(0·00, 2·43)	(0, 68·6)	(0, 30·0)	(44·2, 60·8)	(0, 43·5)	(2·6, 21·9)
n	2	5	2	2	3	9
I_{RT} (mg g^{-1})	18·3 ± 29·0	15·0 ± 21·2	52·5 ± 11·8	19·3 ± 22·1	10·3 ± 6·4	27·8 ± 12·4
	(0, 68·6)	(0, 30·0)	(44·2, 60·8)	(0, 43·5)	(2·6, 21·9)	(6·2, 43·8)
n	5	2	2	3	9	9
I_R (mg g^{-1})	11·6 ± 21·3	17·3 ± 25·3	27·3 ± 26·5	32·6 ± 24·7	11·7 ± 11·5	16·1 ± 14·1
	(0, 68·6)	(0, 68·6)	(0, 60·8)	(0, 60·8)	(0, 43·5)	(0, 62·8)
n	11	7	5	5	13	36
R	0·0106	0·0047	−0·0090	−0·0217	0·0031	0·0120
t (h)	576	600	528	816	840	648
C_{24} (mg g^{-1})	3·67	1·89	−3·85	−25·4	0·90	5·93

n, sample size.

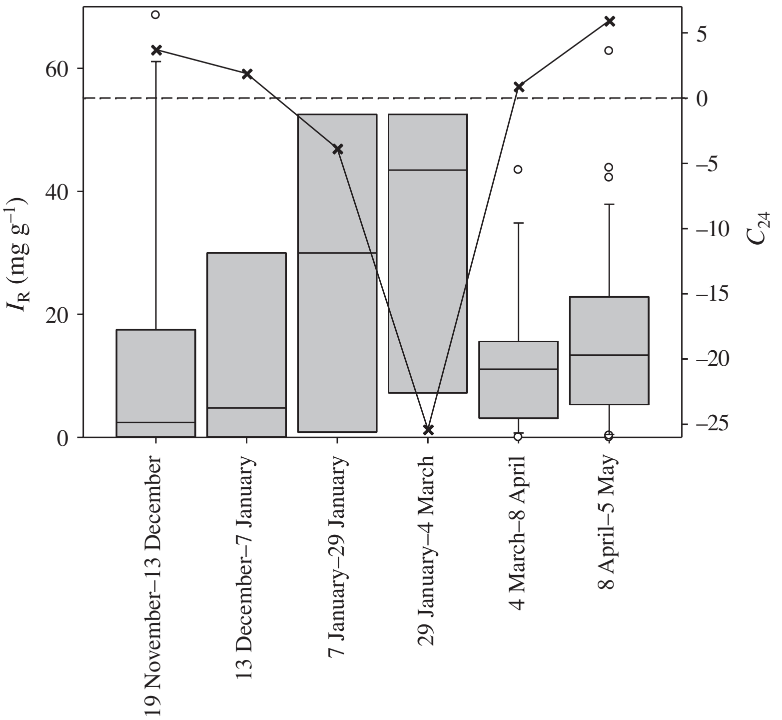

FIG. 7. Relationship between stomach fullness indices (I_R; ▨) and daily consumption rates (C_{24}; ×), calculated for sea-run *Salvelinus fontinalis* captured within intervals from 19 November to 5 May 2010–2012, in Antigonish Harbour, Nova Scotia, Canada. ○, I_R outliers.

lipid deposition (Rikardsen, 2004; Olsen *et al.*, 2006; Rikardsen *et al.*, 2006). Considering members of the genus *Salvelinus* display a high degree of life-history variability, with anadromous populations exhibiting irregular periods of marine residence (Brenkman *et al.*, 2007), it is unclear which of these strategies is more common for *S. fontinalis*. *Salvelinus fontinalis* feeding in Antigonish Harbour from autumn to spring appear to channel energy gained into somatic growth. An individual (258 mm L_F) tagged on 7 April in Antigonish Harbour was recaptured on 14 June in the Wrights River, and revealed an L_F increase of 47 mm in 68 days (unpubl. data), lending evidence for rapid somatic growth. Significant increases in body mass and condition from autumn to spring, however, also suggested allocation to lipid deposition. More detailed studies would be needed to determine if energy allocation changes with ontogeny and season (Olsen *et al.*, 2006).

SEASONAL DIET

Amphipods have been a dominant food item during *Salvelinus* spp. marine migrations (Gaudreault *et al.*, 1982; Morinville & Rasmussen, 2006; Spares *et al.*, 2012). Gaudreault *et al.* (1982) found crustaceans (75% Gammaridae and 25% Mysidacea) to occur in *S. fontinalis*'s diet throughout their entire marine migration in the St-Jean Estuary, Québec, consuming the most during January to February (75·0 %*O* and 94·7 %*N*).

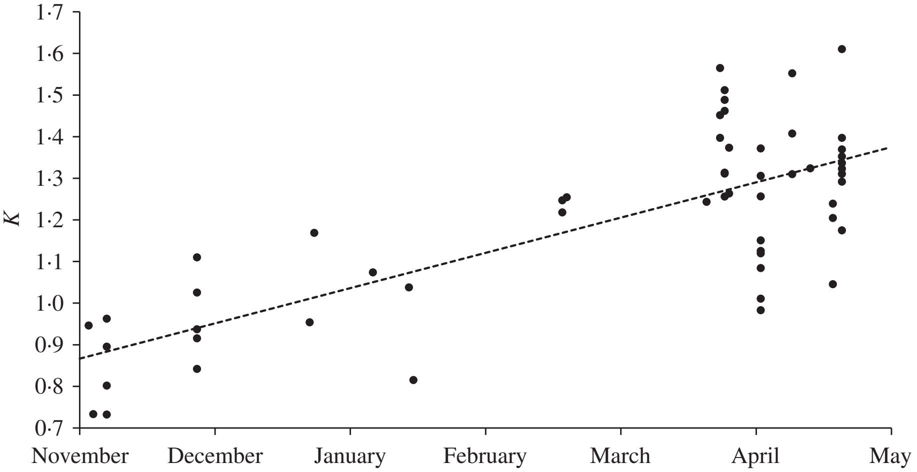

FIG. 8. Body condition factor (*K*) of sea-run *Salvelinus fontinalis* captured in Antigonish Harbour, Nova Scotia, Canada, from November to May 2010–2012 ($y = 0{\cdot}0032x - 129{\cdot}87$; $r^2 = 0{\cdot}536$, $n = 63$).

In contrast, Antigonish Harbour migrants consumed no Gammaridae during winter, and the most during autumn (71·4 %*O*, 84·5 %*N* and 74·6% I_{RI}). Migrants in the Matamek River estuary, Québec, similarly consumed amphipods the most from late-August to early October (Whoriskey *et al.*, 1981). Considering amphipods breed during summer (Ruppert & Barnes, 1994), their increased abundance during late-summer and autumn due to recruitment was probably taken advantage of by feeding *S. fontinalis*.

Nereidae were considered a major winter diet component (14·3 %*O*, 45·0 %*N* and 11·9 %*M*) of Antigonish Harbour *S. fontinalis*, but analyses were based on nine *S. fontinalis*, with one consuming 36 *Nereis* sp. and one *Fundulus* sp., thus, possibly misrepresenting the importance of Nereidae. Gaudreault *et al.* (1982) found winter polychaete predation to be considerably lower (7·1 %*O* and 1·5 %*N*, $n = 28$). Occurrence of Nereidae in this study coincided with the highest percentage of vegetal and miscellaneous matter in stomachs (43%), suggesting *S. fontinalis* were cued to benthic feeding. Cueing on nektobenthic and benthic prey during winter, mainly crustaceans and polychaetes, has been observed in other *S. fontinalis* (Gaudreault *et al.*, 1982) and *S. trutta* (Pemberton, 1976; Rikardsen *et al.*, 2006) populations. Morinville & Rasmussen (2006) noted polychaetes as a dominant prey of *S. fontinalis* remaining near freshwater inputs of St Marguerite Bay, and suggested that estuarine fidelity minimized osmoregulation costs. This may be the case for Antigonish Harbour *S. fontinalis*, as most of them were captured during winter in the estuary of the West River where mean ± s.d. salinities and water temperatures were 4·5 ± 6·9 (minimum = 0·1, maximum = 21·5; $n = 57$) and 1·0 ± 1·4° C (minimum = −0·1° C, maximum = 6·1° C; $n = 57$), respectively (unpubl. data). Ice fishers provided information on observations of *S. fontinalis* feeding on polychaetes in shallow water (depth < 1 m) near the main river channel during February and March. This was further supported by sea-run *S. trutta*, caught concurrently with *S. fontinalis*, regurgitating *Nereis* sp. only (A. D. Spares, pers. obs.). Whether or not the present results accurately estimate the contribution of Nereidae to Antigonish Harbour *S. fontinalis* winter diet, the fact that Nereidae do not occur as prey in autumn, and are more important in winter compared to spring, suggests a seasonal specialization.

Insects occurred as prey during the entire sampling period and included 20 taxa, but their occurrence was limited to a few stomachs. For example, two *S. fontinalis* captured in spring had consumed 13 and 15 taxa, of which 11 (85%) and 14 (93%), respectively, were insects. Most insect prey during spring were Carabidae and Scarabaeidae, probably taken as surface prey (Rikardsen *et al.*, 2006) considering terrestrial ecosystems virtually enclose Antigonish Harbour (Fig. 1). Aquatic insect prey occurred in all seasons, which suggested individuals had returned to fresh water briefly, or had re-entered salt water from an extended freshwater stay. The caddisfly tube found in an individual captured on 5 March may have been undigested and unevacuated for weeks considering cold water temperatures and the tube's sand grain composition (A. D. Spares, pers. obs.). Gaudreault *et al.* (1982) found Ephemeroptera larvae and Trichoptera during January to February in *S. fontinalis* stomachs, which suggested migrants were foraging near freshwater inputs. Aquatic insects occurring in early May samples also suggests that individuals may have re-entered fresh water temporarily before their main re-entry during mid-May to mid-June for summer and autumn residency (unpubl. data). The occurrence of aquatic insects from autumn to spring may suggest site-fidelity to freshwater inputs (Spares *et al.*, 2012).

Prey diversity, indicated by *n* of taxa consumed by migrants, varied considerably between a winter minimum ($n = 5$) and spring maximum ($n = 32$). This relative diet difference probably related to increased terrestrial insect activity in spring, which increased abundance of optimally sized prey items (Morinville & Rasmussen, 2006). The median *n* of taxa consumed per stomach remained the same throughout all seasons.

In fresh water, salmonids must often attain a size of 150–250 mm to show piscivory; however, this minimum size threshold varies by species. *Salmo trutta*, *S. fontinalis*, bull trout *Salvelinus confluentus* (Suckley 1859), cutthroat trout *Oncorhynchus clarki* (Richardson 1836) and sockeye salmon *Oncorhynchus nerka* (Walbaum 1792) have been shown to employ piscivory at L_F >200 mm (Ibbotson *et al.*, 1996; Moyle, 2002; Nowak *et al.*, 2004; Miller *et al.*, 2007; Browne & Rasmussen, 2009). Piscivory, however, has been observed in an individual *S. trutta* as small as 85 mm L_F (Sánchez-Hernández *et al.*, 2012). *Salmo salar* smolts at L_F <150 mm have been documented feeding extensively on fish larvae upon sea entry, suggesting prey type and availability influence piscivorous behaviour (Rikardsen & Dempson, 2011).

The present results concur with the general rule that anadromous *S. fontinalis* >250 mm L_F show piscivory (Dutil & Power, 1980; Whoriskey *et al.*, 1981; Gaudreault *et al.*, 1982; O'Connell, 1982; Morinville & Rasmussen, 2006). The few *S. fontinalis* Gaudreault *et al.* (1982) found feeding on fish in the upper St-Jean Estuary during winter had focused on Gasterosteidae (75% *Pungitius* sp. and 25% *Gasterosteus* sp.), yet more *S. fontinalis* switched to piscivory during spring (May to June), adding *Ammodytes* sp. and *Osmerus* sp., and had fishes dominating diets by summer as migrants moved into the lower estuary (July to August; 45·5 %*O* and 50 %*N*). St Marguerite Bay *S. fontinalis* preyed almost entirely on *Osmerus* sp. in the upper Saguenay River from winter to spring, but diversified to also include Fundulidae, *Ammodytes* sp. and Gasterosteidae in St Marguerite Bay from May to September (Morinville & Rasmussen, 2006). Acoustic tracking and stomach content analysis of *S. fontinalis* in Sheet Harbour, Nova Scotia, revealed migrants remained around shoals to feed on spring-spawned Atlantic herring larvae *Clupea harengus* L. 1758 (E. A. Halfyard, pers. comm.). Antigonish Harbour *S. fontinalis* followed a similar pattern, focusing almost entirely on Fundulidae and Gasterosteidae from autumn to spring in the inner harbour, but demonstrated seasonal diet changes to include Atherinopsidae during winter and spring and Anguillidae during spring. *Menidia menidia* occurred only during winter and spring, agreeing with their overwintering under estuarine sea ice (Needler, 1940) and spawning in spring (Scott & Scott, 1988). Antigonish Harbour *S. fontinalis* appear to use piscivory as their dominant feeding strategy, and similar to other populations (Dutil & Power, 1980; Whoriskey *et al.*, 1981; Gaudreault *et al.*, 1982; O'Connell, 1982; Morinville & Rasmussen, 2006) appear to focus on one or two prey types depending on habitat or season.

The Shannon–Weiner diversity index (H') was similar to stomach content analysis results, being the highest in autumn (1·28) and lowest in spring (1·06), which suggested that fish foraged on multiple taxa, with relative differences possibly relating to *S. fontinalis* feeding on both freshwater and saltwater prey during their return from rivers to the harbour in autumn, and their increasing preference for Fundulidae and Gasterosteidae fishes over the entire estuarine residency relating to the lowest diversity in spring. High preference for a few fishes during the entire migration is further indicated by complete seasonal diet overlap using Schoener's overlap index. Decreasing H' from autumn to spring seemingly contradicts the total number of taxa present within seasonal samples, which showed the lowest diversity of prey during winter and highest in spring, but H' is largely calculated using %M, which further reinforces the importance of a few taxa in Antigonish Harbour *S. fontinalis* diet.

SEASONAL FEEDING INTENSITY

All season samples of Antigonish Harbour *S. fontinalis* had some empty stomachs (<22% ES) similar to actively feeding *S. fontinalis* during winter in the upper Saguenay River, Québec (<20% ES; Morinville & Rasmussen, 2006). Indeed, none of the seasonal or overall % ES were even close to *S. fontinalis* which had ceased feeding (>75% ES; Morinville & Rasmussen, 2006). Higher occurrence of empty stomachs in autumn (18%) and winter (22%) compared to spring (7%), however, suggested that fish may have been having difficulty in feeding during autumn and winter. It should be noted that low sample sizes for autumn and winter may have misrepresented actual percentages. The present results agreed with sea-run *S. trutta* which showed that stomach fullness and feeding rates were lowest in autumn, along with a high percentage of empty stomachs (45–48%; Rikardsen *et al.*, 2006).

The lowest median M of prey and median stomach fullness index of Antigonish Harbour *S. fontinalis* during autumn indicates a low feeding intensity. Gaudreault *et al.* (1982) suggested that the low mean stomach fullness index for *S. fontinalis* captured in fresh water was due to low prey availability. This may suggest that some Antigonish Harbour *S. fontinalis* had recently re-entered salt water from fresh water during autumn and were experiencing difficulty in obtaining prey. Mean stomach fullness indices were virtually the same for autumn (11·6 mg g^{-1}) and spring (11·7 mg g^{-1}), suggesting similar feeding intensities. Autumn and spring indices were comparable to indices (16·7–18·1 mg g^{-1}) for *S. fontinalis* feeding in the St-Jean Estuary from winter to summer (Gaudreault *et al.*, 1982). The difference between autumn median and mean stomach fullness indices for Antigonish Harbour *S. fontinalis* was associated with a low sample size, yet the spread of data agreed with individuals exhibiting varying degrees of feeding intensity at the beginning of their marine migration.

The present winter results appear to suggest that feeding intensity peaked with the highest median prey M and median and mean stomach fullness indices, but these results were contradicted by the highest % ES and negative consumption rates. Other *S. fontinalis* seasonal feeding studies have found higher M_P in winter (Elliott & Jenkins, 1972) and similar stomach fullness indices in winter and summer (Gaudreault *et al.*, 1982). No mid-winter sampling was conducted by Rikardsen *et al.* (2006); however, consumption rates for *S. trutta* were decreasing from November to December and increasing from March to April, which suggests a minimum consumption rate during mid-winter (January to February) when temperatures approached 1° C (Rikardsen, 2004). This appeared to be the case in this study where negative consumption rates corresponded to the coldest *S. fontinalis* body temperatures, which may have essentially lowered feeding (Power, 1980) and decreased digestion to ineffective rates (Gaudreault *et al.*, 1982; Sweka *et al.*, 2004).

Gastric evacuation periods of *S. fontinalis* indicate that 1 week is needed to empty a full stomach at 4° C (Sweka *et al.*, 2004). Comparing water temperatures at 20 *v.* 4 and 3° C for common roach *Rutilus rutilus* (L. 1758) and *O. nerka*, digestion duration increased three and eight times longer, respectively (Brett & Higgs, 1970; Hofer, 1979). Assuming these rates for *S. fontinalis*, gastric evacuation of a full stomach at 3° C would take 5 weeks. Furthermore, gastric evacuation rate is zero or lower at temperatures $\leq 2{\cdot}8°$ C according to the model proposed by Sweka *et al.* (2004). Evidence of little to no digestion was further supported by no significant difference in the median number of prey items consumed per season and no unidentified (*i.e.* partially digested) fish remains in winter samples.

No significant difference in the median number of prey items consumed per season also suggested continued feeding during winter, further supported by migrants actively taking ice fishing bait (A. D. Spares, pers. obs.). It appeared that low body temperatures were not negatively influencing feeding as much as digestion (Spares *et al.*, 2012). Considering that *S. fontinalis* have been shown to have lower gastric evacuation rates compared to other salmonids (Sweka *et al.*, 2004), Antigonish Harbour *S. fontinalis* may have filled their stomachs, but not emptied them during winter.

Spring median M_P and median and mean stomach fullness indices were between autumn and winter or similar to autumn indices, indicating feeding was the most consistent during this season. Although spring and autumn mean body temperatures were the same (9° C), consumption rates were comparatively higher in spring, suggesting high feeding intensity and prey availability (total n prey taxa = 32). Sea-run *S. trutta*'s consumption rates were the highest from spring to autumn (Rikardsen *et al.*, 2006). Mean stomach fullness was highest for *S. fontinalis* in St-Jean Estuary during spring (Gaudreault *et al.*, 1982), further suggesting optimum feeding conditions.

The significant increase in mean L_F, M and K from autumn to spring revealed that Antigonish Harbour *S. fontinalis* were increasing and maintaining body condition during this period. Mean K values indicated that autumn *S. fontinalis* were in very poor condition compared to migrants captured in spring, with 16% of spring specimens considered trophy class (Barnham & Baxter, 2003). Even the maximum autumn K of 1·11 was considered a poor to fair fish (Barnham & Baxter, 2003). Poor autumn K suggested that migrants had recently re-entered salt water following fasting during summer-autumn in fresh water (Morinville & Rasmussen, 2006). Only two spring

S. fontinalis (5·4%) had $K \leq 1{\cdot}00$, representing poor specimens (Barnham & Baxter, 2003), which may have been individuals recently entering the harbour following overwintering in fresh water (Morinville & Rasmussen, 2006). Most spring specimens (81%) had $K \geq 1{\cdot}20$, which are considered fair to excellent fish (Barnham & Baxter, 2003). Considering that many spring migrants (37%) were caught from 5 to 10 April, at least one more month of feeding was possible, subsequently increasing K even more before migrants began returning to fresh water in early May (unpubl. data).

SOURCES OF ERROR

Antigonish Harbour *S. fontinalis* are part of a special trout management area, and are heavily fished during a limited spring recreational fishery (MacMillan & Madden, 2007). Therefore, sample collection was highest during spring, with autumn and winter sampling restricted to fewer specimens to reduce negative effects on the population (Anon., 2006). It is acknowledged that low autumn and winter sample sizes represent a potentially large source of error, especially comparing seasonal feeding intensity indices and diet differences, where one or two stomachs containing high numbers of different prey could affect summary analyses. For this reason, outliers identified by Systat were removed to allow similar variances to be used for seasonal comparisons of stomach fullness, number of prey consumed and vegetal/miscellaneous mass per stomach. The cumulative taxa abundance curve showed that the maximum *n* of taxa consumed during autumn and winter were reached with relatively few stomachs, thus diet diversity comparisons should be unaffected by the low sample sizes.

Although fishing occurred, no samples were obtained during February, which potentially missed mid-winter trends in diet and feeding intensity. Considering samples were collected in late-January and early March, however, it is observed that migrants behaved similarly during the four-week void. Gaudreault *et al.* (1982) sampled *S. fontinalis* during February and found similar feeding patterns as spring, with prey selection focussing more on invertebrates than fishes. Other seasonal studies have had difficulty in obtaining mid-winter samples (Rikardsen *et al.*, 2006), possibly due to low movement rates of migrants (unpubl. data).

Use of 5·1 cm stretched mesh gillnets and hooks baited with live *Fundulidae* during sampling may account for larger sizes of *S. fontinalis* captured. Fyke nets were only used at the head of the tide, thus smaller fish caught in these may have inhabited the transition zone between fresh and marine waters. The resulting low sample size of smaller *S. fontinalis* (<250 mm L_F) may underestimate the importance of fishes in their diet.

Temperature, type and size of prey, predator size and prior starvation have all been shown to influence gastric evacuation rates in salmonids (Elliott, 1972). Sweka *et al.* (2004) used *S. fontinalis* approximately half the length of individuals in this study; however, they suggested that predator size would not influence gastric evacuation rates for *S. fontinalis* >152 mm $L_{\rm F}$. Elliott (1972) found that predator size did not affect gastric evacuation rates in *S. trutta* ranging from 200 to 300 mm $L_{\rm F}$, thus predator size effect may have been a minimal source of error in the present calculations.

Elliott (1991) showed that *S. trutta* gastric evacuation rates of Gasterosteidae were lower than of amphipods, similar to caddisfly larvae and higher than beetle larvae. Based on these data (Elliott, 1991) and Antigonish Harbour *S. fontinalis* prey (which included amphipods, caddisfly larvae and Gasterosteidae), actual evacuation rates may have been higher than the present estimates using the Sweka *et al.* (2004) beetle and fly larvae prey model. No gastric evacuation model for piscivorous *S. fontinalis* exists, thus Sweka *et al.* (2004) was the most appropriate model for present estimates. Other consumption rate studies for fishes have used the entire digestive tract as intestine evacuation rates are lower (Héroux & Magnan, 1996). As this study used stomach contents only, estimated consumption rates may be overestimates.

ESTUARINE OVERWINTERING

Although there are benefits for anadromous salmonids overwintering in salt water, such as increased abundance of prey, stable water levels and less hazardous ice conditions (Rikardsen *et al.*, 2006), there are associated costs such as increased osmoregulation stress and possible predation (Jensen & Rikardsen, 2008, 2012). Yet, despite sea-run *S. fontinalis* experiencing Antigonish Harbour's winter extremes, such as a minimal body temperatures of −0·5° C and maximum salinity of 22·2 (unpubl. data), migrants maintained and increased body condition from November to May. Other studies have documented *S. fontinalis* overwintering in salinities ranging from 0 to 22 depending on the tide (Castonguay *et al.*, 1982; Gaudreault *et al.*, 1982; Morinville & Rasmussen, 2006). Even spring migrants within Laval Bay, Québec, tolerated a minimum temperature of 6° C in salinities reaching 34 (Curry *et al.*, 2006). Other studies (Jensen & Rikardsen, 2008, 2012) have shown *Salvelinus* sp. overwintering in full-strength sea water during winter, contradicting views that the genus is a poor osmoregulator at low temperatures (Saunders *et al.*, 1975; Finstad *et al.*, 1989; McCormick, 1994; Pennell & Barton, 1996; Claireaux & Audet, 1999). Based on diet and fishing locations, Antigonish Harbour *S. fontinalis* remained in the estuary all winter, frequently moving with the flooding tide to the head of tide mark (A. D. Spares, pers. obs.). This behaviour probably aided osmoregulation at colder temperatures (Morinville & Rasmussen, 2006).

Saltwater feeding from autumn to spring represented a valuable and important source of energy, especially for fish whose condition factor significantly decreased after freshwater fasting and spawning (Morinville & Rasmussen, 2006; Rikardsen *et al.*, 2006). Although Antigonish Harbour *S. fontinalis* diet varied with the seasons, migrants relied on a staple diet of Fundulidae and Gasterosteidae, with seasonal supplements of Gammaridae and Nereidae. Piscivory occurred in *S. fontinalis* >250 mm L_F; however, no predation on *Osmerus* sp. was observed, perhaps due to large prey size (A. D. Spares, pers. obs.). Stomach fullness was expected to be the lowest during winter, but the opposite was observed, which suggested that migrants continued to feed although digestion virtually ceased. It is concluded that Antigonish Harbour *S. fontinalis* undertake an autumn to spring feeding migration, during which migrants continuously maintain and increase body condition. Such a life-history strategy represents an alternative to the more common strategy of overwintering and fasting in fresh water until spring.

Our gratitude goes out to the Nova Scotia Department of Fisheries and Aquaculture (Inland Fisheries Division, Pictou) and Ocean Tracking Network Canada for in-kind equipment and logistical support. Special thanks to the following field volunteers: J. Power, D. MacDonald, E. Halfyard, A. Parker, T. MacFarlane S. M. MacDonald, L. Manthey, S. Andrews, D. MacNeil, K. MacAulay, G. Doucet, M. Namba, J. Bond, K. Partridge, J. S. Kirkchoff, Antigonish Town and County Rivers Association and all fishers who donated samples. Thanks to K. Hillier, M. McLean, D. Fife, L. Cooper and A. Loder for help with prey identification and analyses. Funding was provided by Antigonish Town and County Rivers Association, Canadian Wildlife Federation Orville Erickson Memorial Scholarships, Freshwater Fisheries Research Cooperative (Inland Fisheries Division NS), NSERC PGS-D scholarship (A.D.S.) and the Canada Research Chair programme (M.J.W.S.).

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Estuaries and Coasts (2011) 34:1262–1277
DOI 10.1007/s12237-011-9377-7

The Growth of Estuarine Resources (*Zostera marina*, *Mercenaria mercenaria*, *Crassostrea virginica*, *Argopecten irradians*, *Cyprinodon variegatus*) in Response to Nutrient Loading and Enhanced Suspension Feeding by Adult Shellfish

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Received: 21 September 2010 / Revised: 7 January 2011 / Accepted: 10 January 2011 / Published online: 27 January 2011

Abstract While many coastal ecosystems previously supported high densities of seagrass and abundant bivalves, the impacts of overfishing, eutrophication, harmful algal blooms, and habitat loss have collectively contributed to the decline of these important resources. Despite improvements in wastewater treatment in some watersheds and subsequent reduced nutrient loading to neighboring estuaries, seagrass and bivalve populations in these locations have generally not recovered. We performed three mesocosm experiments to simultaneously examine the contrasting effects of nutrient loading and historic suspension-feeding bivalve densities on the growth of eelgrass (*Zostera marina*), juvenile bivalves (northern quahogs, *Mercenaria mercenaria*; eastern oysters, *Crassostrea virginica*; and bay scallops, *Argopecten irradians*), and juvenile planktivorous fish (sheepshead minnow, *Cyprinodon variegatus*). High nutrient loading rates led to significantly higher phytoplankton (chlorophyll *a*) levels in all experiments, significantly increased growth of juvenile bivalves relative to controls with lower nutrient loading rates in two experiments, and significantly reduced the growth of eelgrass in one experiment. The filtration provided by adult suspension feeders (*M. mercenaria* and *C. virginica*) significantly decreased phytoplankton levels in all experiments, significantly increased light penetration and the growth of eelgrass in one experiment, and significantly decreased the growth of juvenile bivalves and fish in two experiments, all relative to controls with no filtration from adult suspension feeders. These results demonstrate that an appropriate level of nutrient loading can have a positive effect on some estuarine resources and that bivalve filtration can mediate the effects of nutrient loading to the benefit or detriment of different estuarine resources. Future ecosystem-based approaches will need to simultaneously account for anthropogenic nutrient loading and bivalve restoration to successfully manage estuarine resources.

Keywords *Zostera marina* · *Crassostrea virginica* · *Mercenaria mercenaria* · *Argopecten irradians* · *Cyprinodon variegatus* · Eelgrass · Seagrass · Clams · Oysters · Eutrophication · Nutrients · Nutrient loading · Aquaculture · Bivalves · Suspension feeders · Mesocosms · Ecosystem-based management · Estuarine restoration

Introduction

Estuaries are home to a variety of valuable living resources. Finfish and shellfish are harvested directly in commercial and recreational fisheries, while seagrass beds are considered of paramount importance as structural habitat for shellfish and finfish in many coastal areas (Heck and Wetstone 1977; Irlandi and Peterson 1991; Beck et al. 2001). Many of the world's estuaries currently support lower abundances of finfish, shellfish, and seagrasses than they did historically due to overfishing (Jackson et al. 2001; Lotze et al. 2006), habitat loss (Orth et al. 2006), eutrophication (Nixon 1995; de Jonge et al. 2002), and harmful algal blooms (Hallegraeff 1993; Gobler et al. 2005; Sunda et al. 2006). As such,

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estuarine management plans are typically focused on combating these harmful processes and restoring living resources (Cloern 2001; Newell 2004; Lotze et al. 2006).

Changes in nutrient loading to estuaries can indirectly influence the growth of marine resource species. High rates of nutrient loading have been associated with increases in pelagic productivity, decreased water clarity, hypoxia, and declines in seagrass growth and abundances (Short et al. 1995; Diaz and Rosenberg 2008; Wall et al. 2008). In response, estuarine management efforts often focus primarily on reducing anthropogenic nutrient loading in an effort to curb the negative effects of eutrophication (Cloern 2001; de Jonge et al. 2002). However, some level of nutrient loading must be necessary to sustain primary and secondary production (Nixon and Buckley 2002). Higher levels of inorganic nutrients can enhance primary production rates and can favor larger phytoplankton cells (Malone 1980; Raven and Kubler 2002), such as diatoms and prymnesiophytes, which are generally considered a good source of nutrition for bivalves (Beukema and Cadee 1991; Wikfors et al. 1992; Weiss et al. 2007). Studies in several estuaries have shown that blue mussels (*Mytilus edulis*), northern quahogs (*Mercenaria mercenaria*), and softshell clams (*Mya arenaria*) can respond positively to increased nitrogen loading and high chlorophyll *a* levels in their habitats (Vanstralen and Dijkema 1994; Weiss et al. 2002, 2007; Carmichael et al. 2004). Weiss et al. (2002) and Carmichael et al. (2004) found that shell growth, soft tissue growth, and survival of *M. mercenaria* and *M. arenaria* increased with nitrogen loading rates along a naturally occurring gradient in Waquoit Bay, MA, USA. They attribute these changes to increased quantity and quality of food particles due to nitrogen enrichment (Carmichael and Valiela 2005), although a similar response has not been found for bay scallops (*Argopecten irradians*, Shriver et al. 2002). While nutrient overloading in estuaries has a well-known set of negative consequences (Valiela et al. 1992; Nixon 1995; Kemp et al. 2005), the stimulation of secondary production in bivalves could be an overlooked positive effect of nutrient loading (Nixon and Buckley 2002; Carmichael et al. 2004; Carmichael and Valiela 2005), especially in shallow ecosystems with well-mixed water columns that rarely experience hypoxia.

As described in many studies and reviews, suspension-feeding bivalves are both a fisheries resource and a provider of key ecosystem services (Dame 1996). These animals can have a variety of effects on estuaries through their suspension-feeding activities, such as reducing phytoplankton biomass and other suspended particles (Officer et al. 1982; Hawkins et al. 1996; Barille et al. 1997), cycling nutrients and biomass between the benthos and the water column (Kautsky and Evans 1987; Smaal and Prins 1993), control of harmful algae (Cerrato et al. 2004), increased light penetration (Newell and Koch 2004), and facilitating the growth of benthic plants (Peterson and Heck 2001; Wall et al. 2008).

As bivalve populations have declined through overfishing, habitat loss, and disease, these ecosystem services have been lost and there are currently few estuaries with natural densities of bivalves sufficient to exert ecosystem-wide effects (Newell 1988; Lotze et al. 2006). In the absence of dense natural bivalve populations, bivalve aquaculture may achieve similar levels of ecosystem-wide impact (Souchu et al. 2001; Dumbauld et al. 2009). Some managers have considered aquaculture as a means to restore ecosystem functions previously provided by natural populations (Newell 2004; Ruesink et al. 2005), to combat eutrophication (Gifford et al. 2004; Cerco and Noel 2007), or to ease harvest pressures on wild populations (Dolmer and Frandsen 2002). Aquaculture is on the rise worldwide, and bivalve aquaculture may avoid some of the pitfalls of finfish aquaculture (Naylor et al. 2000) while controlling phytoplankton blooms and affecting carbon and nutrient cycling in ways that are comparable to natural shellfish populations (Smaal et al. 2001; Newell 2004; Huang et al. 2008).

Commercial bivalve aquaculture operations strive to grow a maximum number of shellfish in a minimum of space (Frechette et al. 1992), with locally high filtration rates sometimes leading to “self-thinning” through density-dependent food limitation (Rheault and Rice 1996; Zhou et al. 2006). It is not well-known how these locally high filtration rates interact with adjacent natural bivalve populations (Ferreira et al. 2008), but locally high biodeposition rates from aquaculture have produced negative effects in some systems (Tenore et al. 1982; Feng et al. 2004), and intense aquaculture can exceed the ecological carrying capacity of some estuaries (Nunes et al. 2003; Duarte et al. 2003). As aquaculture develops for both commercial and restoration purposes, an improved understanding of these effects will help managers use bivalves to achieve healthy ecosystem functions (Dumbauld et al. 2009).

This study was designed to examine the combined effects of nutrient loading and adult bivalve filtration on the growth and survival of estuarine resource species: juvenile northern quahogs (*M. mercenaria*), bay scallops (*A. irradians*), and oysters (*Crassostrea virginica*); a juvenile planktivorous fish (sheepshead minnow, *Cyprinodon variegatus*); and eelgrass (*Zostera marina*). Juvenile sheepshead minnows are known to feed on both zooplankton and large phytoplankton (Samson et al. 2008). These five species were placed into an array of mesocosms with treatments of high or low nutrient loading and presence or absence of adult bivalves arranged in a 2×2 factorial design. The growth of all populations along with levels of light and size-fractionated chlorophyll *a* was monitored during three experiments which demonstrated that both nutrient loading and adult bivalve filtration can strongly influence the growth of multiple estuarine resources.

Methods

We conducted three experiments with mesocosms placed in eastern Shinnecock Bay at the Stony Brook-Southampton Marine Science Center from June 5, 2007 to September 6, 2007. Shinnecock Bay is part of Long Island's south shore estuary lagoons (NY, USA) which have followed a trajectory in the decline of resources common to many estuaries around the world (Bricelj and Kuenstner 1989; McHugh 1991; Gobler et al. 2005). Specifically, these lagoons have seen declines in shellfish such as the hard clam (a.k.a. northern quahog, McHugh 1991), the bay scallop (Bricelj and Kuenstner 1989), various finfish, and eelgrass beds (Dennison et al. 1989). The 300-L mesocosms used in this study have been utilized previously to yield realistic growth rates and conditions for planktonic communities, seagrass, and shellfish (Cerrato et al. 2004; Wall et al. 2008). The depth of the mesocosms (1.2 m) is within the range of the mean depths found among Long Island's south shore estuary lagoons (Wilson et al. 1991). Moreover, the placement of the tanks in eastern Shinnecock Bay allowed for ambient light and temperature to be maintained during experiments. Replicate experimental mesocosms (n=4 for each treatment) were stocked with juvenile northern quahogs (~10 mm shell length), bay scallops (~10 mm shell height), and/or eastern oysters (~10 mm shell height) at stocking densities (10–20 tank^{-1} or 36–72 m^{-2}; Table 1) more than an order of magnitude lower than standard commercial aquaculture stocking densities (~500 individuals m^{-2}; Barber and Davis 1997; Kraueter and Castagna 2001) to avoid inter- and intraspecific competition for food (Rheault and Rice 1996; Kraueter and Castagna 2001) among juvenile shellfish. Indeed, our estimated community clearance rates of juvenile bivalves indicated they filtered 0.4–1.5% day^{-1} of the total mesocosm volumes. All juvenile bivalves were placed in mesh cages (2 mm mesh size) near the bottom of the mesocosms. Juvenile shellfish were obtained from the Cornell Cooperative Extension shellfish hatchery in Southold, NY, USA. Three-week-old sheepshead minnows (10–15 mm) were obtained from Cosper Environmental Services in Bohemia, NY, USA. These planktivorous fish (Samson et al. 2008) were held in mesh baskets suspended near the tops of the experimental tanks (n=10). A laminar circulating pump (Rio 180®) was utilized to ensure mesocosms were well-mixed. In addition to the suspension feeders, individual shoots of eelgrass (n=16) were transplanted into planters containing low-organic sand and placed in each mesocosm (Wall et al. 2008).

Mesocosms were filled with eastern Shinnecock Bay water during high tide. Water from this region is fairly mesotrophic with mean total N (dissolved+particulate) concentrations of 0.2±0.1 mg N L^{-1} or 16±8 μM N measured from 2000 to 2005 (n=50 measurements; SCDHS 2000–2005). For each experiment, we established a low nutrient loading rate for half of the experimental tanks (DIN loading of 0.065–0.255 mmol N m^{-2} day^{-1}) using a 1–2% day^{-1} exchange with Shinnecock Bay water. The other half of the tanks received a high nutrient loading rate (5.49–10.70 mmol N m^{-2} day^{-1}) that reflected ambient exchange plus nutrient additions of ammonium and the Redfieldian equivalent (16:1) of orthophosphate. These nutrient loading rates were within the range found in more eutrophic Northeast US estuaries such as the Childs River, MA and Moriches Bay, NY (Taylor et al. 1999). Nutrient stocks were filter-sterilized (0.2 μm) and stored frozen. Experiments were run in semi-continuous mode, with 1–2% of the water volume being replaced daily mimicking the natural slow tidal exchange which occurs in the back-bay regions of the Peconic Estuary and Great South Bay, Long Island, NY, USA, resulting in residence times on the order of 2 to 3 months (Hardy 1976; Wilson et al. 1991). For each experiment, half of the experimental tanks contained adult suspension feeders (northern quahog or eastern oyster) and half of the tanks contained no adult suspension feeders. Stocking densities of adult bivalves in the experimental tanks (21–43 individuals m^{-2}) were comparable to historic densities of shellfish in Long Island South Shore Estuaries (Kassner 1993) but higher than current densities (0–5 individuals m^{-2}; Weiss et al. 2007). Shellfish densities in the experiment treatments were also orders of magnitude lower than stocking densities in modern aquaculture operations (Rheault and Rice 1996). Adult clams measured 56.70±1.18 mm shell length and weighed 1.64±0.11 g ash-free dry weight (AFDW). Adult oysters measured 59.17±0.79 mm shell height and weighed 0.66±0.05 g AFDW. Adult shellfish were locally caught and obtained from seafood markets. The feeding activity of adult shellfish was estimated with a clearance rate method (Riisgard 2001) using water (>15 μg L^{-1} chlorophyll a) from the experimental tanks. Clearance rates were calculated according to the equation:

$$\text{clearance rate} = V/t \times [\ln(\text{chla}_0/\text{chla}_t)]$$

where V is the volume of the container, t is the time, and chl a_0 and chl a_t are the chl a levels at the initial reading and at time t, respectively. This measurement was performed once per species. A "community" clearance rate was estimated from these data using the average individual clearance rate and the number of individuals in the tank. An estimated clearance rate for the entire tank volume to be processed by the adult shellfish was calculated for each tank by dividing

Table 1 Stocking densities of response organisms and summary of experimental conditions

			Experiment 1	Experiment 2	Experiment 3
Stocking densities of response organisms (n=# per tank)	Juvenile bivalves	*M. mercenaria*	10	20	
		C. virginica	15	10	10
		A. irradians	0	0	10
	Juvenile fish	*C. variegatus*	0	0	10
	Eelgrass shoots	*Z. marina*	16	16	16
Experimental conditions	Adult bivalve species		*M. mercenaria*	*C. virginica*	*M. mercenaria*
	Density of adult bivalves	+ Bivalves	29 m^{-2}	21 m^{-2}	43 m^{-2}
		− Bivalves	0	0	0
	Estimated clearance rate of tank volume from + bivalves treatment		42% day^{-1}	67% day^{-1}	63% day^{-1}
	Exchange with ambient water		1% day^{-1}	2% day^{-1}	2% day^{-1}
	Nutrient loading rate (mmol N m^{-2} day^{-1})	High N	10.70	5.75	5.49
		Low N	0.065	0.255	0.134

Treatments were "+ bivalves" or "− bivalves" for presence or absence of adult bivalves and "high N" or "low N" for high or low nutrient loading. Nutrients were added as 16:1 inorganic N/P. A total of 16 tanks were used for each 2×2 factorial experiment with n=4 tanks per treatment combination

this community clearance rate by the tank volume. A summary of experimental conditions for all three experiments is presented in Table 1.

Experiments were conducted for ~2 weeks, and shellfish growth was assessed via the changes in AFDW of tissue or by changes in shell lengths between initial and final individuals within each mesocosm (Weiss et al. 2007). The length of juvenile clams was measured by shell length (anterior-posterior; Kraueter and Castagna 2001), and the size of juvenile oysters and scallops was measured by shell height (hinge-ventral margin; Rheault and Rice 1996). Bivalve tissue was dried at 70°C for at least 24 h and then ashed at 450°C for an additional 4 h (Gabbott and Walker 1971; Bass et al. 1990). One hundred bivalves of each species were selected from the initial set to provide a mean initial tissue AFDW. When fewer than 100 individuals were available for a mean initial AFDW, initial AFDW's were hind-casted based on initial lengths using length-weight regressions from 100+ individuals of the same species and size class. Juvenile fish growth was measured by total length only. Mean growth rates for all species based on length or weight were calculated by the change in length or tissue AFDW divided by the number of days between initial and final measurements. The quality and quantity of phytoplankton food particles available for bivalves was assessed by measuring whole and size-fractionated chlorophyll *a* (>5 µm) using polycarbonate filters and standard fluorometric techniques (Parsons et al. 1984). Chlorophyll in the <5-µm-size fraction was calculated as the difference between whole and >5 µm chl *a*. Additional whole water samples were collected on pre-combusted glass fiber filters for the analysis of particulate organic carbon (POC) and nitrogen (PON) on a CE Instruments Flash 1112 elemental analyzer (Sharp 1974).

Experimental treatment effects on eelgrass productivity and epiphyte biomass were assessed by marking then harvesting eelgrass shoots from each replicate mesocosm. Leaf production during the experiment was measured using a modified leaf marking technique (Ibarra-Obando and Boudouresque 1994). Sixteen eelgrass shoots were marked at the base of the leaves by driving an 18-gauge hypodermic needle through all of the leaves on the shoot. The marked shoots were allowed to grow for the length of the experiment (13–15 days), after which all above-ground leaf material was harvested. In the laboratory, daily gross above-ground productivity and leaf epibiont biomass (milligrams of AFDW per square centimeter leaf area) was determined. Productivity was determined by both mass (milligrams per shoot per day) and leaf area growth (square centimeters per shoot per day). Epiphyte biomass was scraped from each leaf, dried for at least 24 h at 70°C, and then ashed at 450°C for an additional 4 h to determine AFDW.

Bottom light levels in each mesocosm were measured every 15 min by HOBO© Pendant-style data loggers with light sensors. A data logger was placed in each experimental tank near the bottom at a depth of approximately 1 m, a height just above eelgrass and shellfish cages preventing the obstruction of incoming light. A mean daily light level for each experimental tank was calculated by averaging values between 10:00 and 14:00 h, when the sun was most directly overhead. Since the HOBO© data loggers measure visible light levels in lux instead of photosynthetically active radiation (PAR) in micromoles per square meter per second, we compared measurement of light with the HOBO© loggers to those obtained with a LiCor© LI-192 underwater quantum sensor of PAR. There was a highly significant linear relationship between visible light in lux as measured by the HOBO© data logger and PAR as measured by the LiCor© sensor over depths of 0.5–2.0 m (visible light in $\mathrm{LUX}=41.407\times\mathrm{PAR}-408.67$, $r^2=0.98$, $p<0.001$). Based on this finding, we believe that experimental light readings from HOBO© data loggers within our mesocosms were representative of the general trends in PAR.

Seawater dilution experiments were conducted to quantify the rates of microzooplankton grazing of micro-algal biomass within the mesocosm tanks (Landry et al. 1995). During each experiment, 5 L of water from each replicate mesocosm within a treatment were pooled into a 20-L carboy for that treatment. Triplicate samples of 100%, 70%, 40%, and 15% experimental dilutions of whole seawater with filtered seawater (0.2 µm) from each carboy were established in 1 L polycarbonate bottles. To ensure nutrient-replete growth during these experiments, nitrate (20 µM) and orthophosphate (1.25 µM) were added to all of the bottles. A set of triplicate controls of whole seawater without nutrients were also established for each grazing experiments (Landry et al. 1995). Micro-algal growth rates (μ) within experimental bottles were quantified using the formula: $\mu=[\ln(B_t/B_\mathrm{o})]/t$, where μ is the net growth rate, B_t is the amount of biomass (chl *a*) present at the end of the experiments, B_o represents the amount of biomass at the beginning of experiments, and t is the duration of the experiment in days. The slope of first-order linear regressions of dilution of seawater (x-axis) and the net growth rates (y-axis) were used to establish grazing mortality rates (Landry et al. 1995).

Statistical Analysis

Differences in the growth of each animal species and eelgrass were assessed by means of two-way analysis of variance (ANOVA), with nutrient loading level and presence/absence of adult bivalves as the two treatment factors using the software SigmaStat 3.5. When a significant effect on the response variables was detected, multiple comparison tests (Tukey's studentized range) were used to test for significant differences between levels within the treatment. Mortality of juvenile bivalves was analyzed using a G test of independence (Sokal and Rohlf 1995). Chlorophyll *a* and light level trends were analyzed with three-way repeated-measures ANOVAs (ANOVARs) where level of nutrient loading and presence/absence of adult bivalves were the between-subjects effects and day was the repeated within-subjects effect. Each mesocosm tank was considered a subject for this analysis, which was conducted using the software Systat 13. In the case of significant interaction effects in the three-way ANOVAR, the variance was decomposed by means of two-way ANOVARs (day × bivalves and day × nutrients). Data that did not meet ANOVA assumptions were $\log(x+1)$-transformed to achieve normality. All statistical results were considered against a significance level of $\alpha=0.05$.

Results

Experiment 1

Three separate mesocosm experiments were carried out using the above methods (Table 1). Experiment 1 ran from June 5 to June 18, 2007. The average temperature in the experimental tanks was 20.30±0.15°C, the average salinity was 26.42±0.04, and the average dissolved oxygen was $6.65\pm0.14\ \mathrm{mg\ L^{-1}}$. The "low nutrient loading" treatment received an average of $0.065\ \mathrm{mmol\ N\ m^{-2}\ day^{-1}}$ and $0.006\ \mathrm{mmol\ P\ m^{-2}\ day^{-1}}$ through a ~1% $\mathrm{day^{-1}}$ exchange with Shinnecock Bay water whereas the "high nutrient loading" treatment received $10.70\ \mathrm{mmol\ N\ m^{-2}\ day^{-1}}$ and $0.671\ \mathrm{mmol\ P\ m^{-2}\ day^{-1}}$. The densities of adult suspension feeders were 29 or 0 northern quahogs $\mathrm{m^{-2}}$ (8 or 0 individual $\mathrm{tank^{-1}}$). The estimated clearance time from bivalve filtration for the experimental tanks with northern quahogs was 42% $\mathrm{day^{-1}}$. All tanks in this experiment were stocked with juvenile clams, juvenile oysters, and eelgrass (Table 1).

In this experiment, the higher nutrient loading rate ($10.70\ \mathrm{mmol\ N\ m^{-2}\ day^{-1}}$) and the absence of adult clams produced significant increases in chlorophyll *a* compared to the low nutrient loading rate ($0.065\ \mathrm{mmol\ N\ m^{-2}\ day^{-1}}$) and the presence of adult clams (29 individuals $\mathrm{m^{-2}}$) over the course of a 13-day experiment (Fig. 1a, b; $p<0.01$ and $p<0.001$ for nutrient and bivalve treatments, respectively, three-way ANOVAR). The level of whole chl *a* within each mesocosm varied significantly by day ($p<0.001$, Fig. 1a, three-way ANOVAR), and there was also a significant day × bivalve treatment interaction ($p<0.01$). When variance in whole chl *a* levels was decomposed with two-way ANOVARs, the addition of bivalves consistently decreased whole chl *a* across both nutrient treatments ($p<0.05$), while nutrient loading significantly increased whole chl *a* only within the bivalve-added treatment ($p<0.05$). Despite consistent directional effects from the nutrient and bivalve treatments (Fig. 1b), chl *a* in the >5-µm-size fraction varied significantly only by day ($p<0.001$, three-way ANOVAR) and not by treatment. Chlorophyll *a* in the <5-µm-size fraction was significantly increased by high nutrient loading

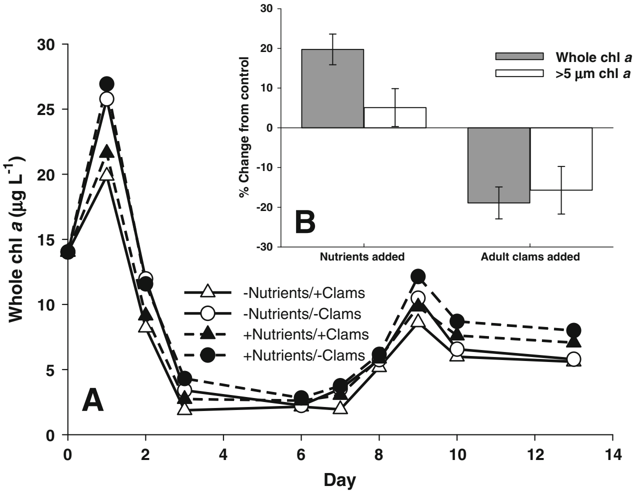

Fig. 1 Chlorophyll *a* dynamics in experiment 1. Time series data points (**a**) represent the mean ($n=4$) for each of the treatment combinations. *Error bars* are not presented for the sake of visual clarity. The mean relative standard deviation of measurements for whole chl *a* was 19.8% during the experiment. *Inset* (**b**) shows mean (± SE) daily percent increase or decrease from nutrient addition over both bivalve treatments and from bivalve addition over both nutrient treatments. See text for magnitudes of nutrient loading and densities of adult bivalves

and decreased by adult clam filtration ($p<0.01$ in both cases, three-way ANOVAR, data not shown). Levels of chl *a* <5 µm also varied significantly by day ($p<0.001$) and day×bivalve treatment interaction ($p<0.05$). When this variance was decomposed with two-way ANOVARs, the addition of bivalves produced a significant drop in <5 µm chl *a* only within the high nutrient loading treatment ($p<0.01$), while nutrient loading produced a significant increase in <5 µm chl *a* only within the bivalve-added treatment ($p<0.05$). The molar ratio of POC/PON was significantly higher under low nutrient loading (9.70±0.61; Table 2) and the absence of adult clams (10.51±0.45) compared to high nutrient loading (9.22±0.26) and the presence of adult clams (8.83±0.24; $p<0.05$ for nutrient treatment, $p<0.01$ for clam filtration treatment, two-way ANOVA).

The highest juvenile clam growth was in the presence of high nutrient loading and in the absence of adult clams, while the lowest was without nutrient loading but with adult clams present (Fig. 2a). However, only the nutrient loading treatment had a statistically significant effect: juvenile clam shell growth (Fig. 2a) and juvenile oyster soft tissue growth (Fig. 2b) were both significantly higher in the high nutrient loading treatment (0.032±0.009 mm day^{-1} and 0.078±0.016 mg AFDW day^{-1}, respectively) compared with treatments without experimental nutrient addition (0.00±0.01 mm day^{-1} and 0.034±0.015 mg AFDW day^{-1}, respectively; $p<0.05$ for each, two-way ANOVA). Despite the changes in chlorophyll *a*, light levels were not significantly different among treatments and subsequently eelgrass growth was not affected by the experimental treatments. Microzooplankton grazing rate data were not available for this experiment.

Experiment 2

Experiment 2 ran from July 12 to July 27, 2007. The average temperature in the experimental tanks was 24.27±0.16°C, the average salinity was 28.02±0.16, and the average dissolved oxygen was 5.83±0.12 mg L^{-1}. The "low nutrient loading" treatment received an average of 0.255 mmol N m^{-2} day^{-1} and 0.072 mmol P m^{-2} day^{-1} through a ~2% day^{-1} exchange with Shinnecock Bay water. The "high nutrient loading" treatment received ambient exchange plus a daily experimental nutrient addition for a total of 5.75 mmol N m^{-2} day^{-1} and 0.416 mmol P m^{-2} day^{-1}. The densities of adult suspension feeders were 21 or 0 eastern oysters m^{-2} (6 or 0 individual tank^{-1}). The estimated turnover from bivalve filtration for the experimental tanks with oysters was 67% day^{-1}. All tanks in this experiment were stocked with juvenile clams, juvenile oysters, and eelgrass (Table 1).

Although both treatments produced consistent directional effects on the levels of whole chlorophyll *a* (Fig. 3a, b), whole chl *a* was not significantly altered by the treatments ($p>0.05$, three-way ANOVAR). Whole chl *a* within each mesocosm tank varied significantly by day (Fig. 3a, $p<0.01$, three-way ANOVAR), and there was also a significant day×bivalve treatment interaction ($p<0.05$). When this variance was decomposed using two-way ANOVARs, this interaction effect was removed and day was the only significant source of variation in whole chl *a*. Similarly, chl *a* in the >5-µm-size class displayed consistent directional effects according to the treatments (Fig. 3b), but the only significant variation was by day ($p<0.001$, three-way ANOVAR). In contrast, chl *a* in the <5-µm-size fraction was significantly enhanced by nutrient loading ($p<0.01$), significantly reduced by the addition of bivalves ($p<0.01$), and displayed a nutrient treatment×bivalve treatment interaction ($p<0.01$, three-way ANOVAR, data not shown). When this variance was decomposed using two-way ANOVARs, the decrease of <5 µm chl *a* by bivalves occurred only within the high nutrient loading treatment ($p<0.05$) and the increase in <5 µm chl *a* by nutrient loading occurred only within the no bivalves treatment ($p<0.01$).

Juvenile clam growth was significantly higher in the high nutrient loading treatment (0.039±0.003 mm day^{-1} and 0.058±0.005 mg AFDW day^{-1}) compared to the low nutrient loading treatment (0.030±0.003 mm day^{-1} and 0.033±0.005 mg AFDW day^{-1}) when measured by shell length (data not shown; $p<0.05$, two-way ANOVA) or by dry tissue weight (Fig. 4a; $p<0.001$, two-way ANOVA). Juvenile clam growth was not affected by the adult oyster filtration treatment. In contrast, the juvenile oysters

Table 2 Levels of chlorophyll *a*, POC, PON, and microzooplankton grazing rates

		Whole chl *a* (µg L^{-1})	>5 µm chl *a* (µg L^{-1})	POC (µM)	PON (µM)	POC/PON	Microzooplankton grazing rate day^{-1}
Experiment 1	Low N/+ bivalves	6.60±0.91	4.92±0.75	244.00±44.06	27.21±3.83	8.85±0.33	No data available
	Low N/− bivalves	8.39±1.18	5.58±0.80	150.46±13.55	13.34±2.09	11.40±0.77	
	High N/+ bivalves	7.72±0.95	5.44±0.83	200.40±26.71	23.13±3.66	8.81±0.39	
	High N/− bivalves	9.38±1.19	6.21±0.84	184.90±15.63	19.25±1.62	9.62±0.23	
Experiment 2	Low N/+ bivalves	3.57±0.29	2.12±0.27	No data available	No data available	No data available	2.36±0.52
	Low N/− bivalves	3.89±0.42	2.83±0.41				2.39±0.63
	High N/+ bivalves	4.64±0.68	2.62±0.43				2.36±0.45
	High N/− bivalves	6.51±0.46	3.68±0.41				2.31±0.53
Experiment 3	Low N/+ bivalves	14.15±2.61	8.96±2.30	120.53±18.73	16.05±2.55	7.65±0.89	0.55±0.32
	Low N/− bivalves	21.76±3.25	22.58±4.23	272.08±15.24	32.77±3.72	8.42±0.54	0.45±0.07
	High N/+ bivalves	19.92±3.64	12.40±3.69	113.24±2.66	16.31±1.07	7.00±0.46	0.73±0.19
	High N/− bivalves	29.00±3.77	31.95±5.76	348.62±9.64	43.95±3.42	8.00±0.47	0.63±0.17

Values are mean ± SE of experimental tanks for each treatment combination averaged over the course of each experiment. Treatments were "+ bivalves" or "− bivalves" for presence or absence of adult bivalves and "high N" or "low N" for high or low nutrient loading. Nutrients were added as 16:1 inorganic N/P. A total of 16 tanks were used for each 2×2 factorial experiment with $n=4$ tanks per treatment combination. Values of >5 µm chl *a* that are greater than whole chl *a* for experiment 3 reflect plankton communities where virtually all chl *a* is in the >5-µm-size fraction

responded to the adult bivalve treatment; juvenile oyster growth was significantly decreased in the presence of adult oyster filtration (Fig. 4b; $p<0.01$; two-way ANOVA) but was not affected by the nutrient loading treatments. Juvenile oyster growth was 0.131±0.022 mg AFDW day^{-1} in the absence of adult oysters and was 0.033±0.017 mg AFDW day^{-1} in the presence of adult oysters. Light levels and eelgrass growth were not significantly affected by the experimental treatments (two-way ANOVA), although epiphyte biomass on eelgrass leaves was significantly higher under high nutrient loading (0.164±0.013 mg AFDW cm^{-2}) and adult oyster filtration (0.179±0.011 mg AFDW cm^{-2}) compared to low nutrient loading (0.140±0.012 mg AFDW cm^{-2}) and no adult oyster filtration (0.126±0.006 mg AFDW cm^{-2}; $p<0.05$ by nutrient treatment, $p<0.001$ by oyster treatment, two-way ANOVA). Microzooplankton grazing rates were not significantly different between treatments and ranged from 2.31 to 2.39 day^{-1} (Table 2). POC/PON data were not available for this experiment.

Experiment 3

Experiment 3 ran from August 22 to September 6, 2007. The average temperature in the experimental tanks was 24.56±0.15°C, the average salinity was 29.73±0.09, and the average dissolved oxygen was 6.16±0.16 mg L^{-1}. The "low nutrient loading" treatment received an average of 0.134 mmol N m^{-2} day^{-1} and 0.099 mmol P m^{-2} day^{-1} through a ~2% day^{-1} exchange with Shinnecock Bay water. The "high nutrient loading" treatment received ambient exchange plus a daily experimental nutrient addition for a total of 5.49 mmol N m^{-2} day^{-1} and 0.434 mmol P m^{-2} day^{-1}. The densities of adult suspension feeders were 43 or 0 clam m^{-2} (12 or 0 individual $tank^{-1}$). The estimated turnover rate from bivalve filtration for the experimental tanks with clams was 63% day^{-1}. All tanks in this experiment were stocked with juvenile scallops, juvenile clams, juvenile oysters, juvenile sheepshead minnows, and eelgrass (Table 1).

In this experiment, the presence of adult northern quahogs (43 individuals m^{-2}) produced significant decreases in total chlorophyll *a* compared to the absence of adult clams over the course of a 15-day experiment ($p<0.001$, three-way ANOVAR, Fig. 5a, b), and whole chl *a* also varied significantly over time within each mesocosm tank ($p<0.001$). The significant decrease of whole chl *a* by the bivalve-added treatment was consistent across both levels of nutrient loading and over time during the experiment (Fig. 5a, b). Even though the high nutrient loading rate (5.49 mmol N m^{-2} day^{-1}) produced a consistent directional effect on whole chl *a* compared to the low nutrient loading rate (0.134 mmol N m^{-2} day^{-1}, Fig. 5b), this effect was not statistically significant ($p>0.05$, three-way ANOVAR). Trends in whole chl *a* were paralleled by the >5-µm-size fraction of chl *a*, which was decreased by the addition of adult bivalves ($p<0.001$, three-way ANOVAR, Fig. 5b) and also varied within each mesocosm tank by day ($p<0.05$). There was also a significant day×bivalve treatment interactive effect on levels on >5 µm chl *a* ($p<0.05$, three-way ANOVAR). When this variance was decomposed using two-way ANOVARs, the interactive effect was removed. Although >5 µm chl *a* was consistently increased by nutrient loading (Fig. 5b), this effect was not statistically significant ($p>0.05$). In contrast to experiments 1 and 2, chl

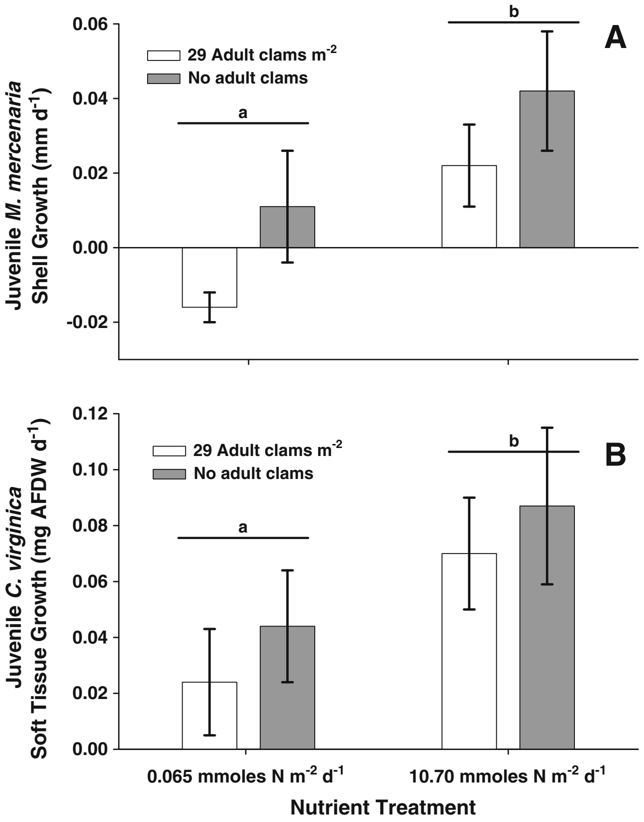

Fig. 2 Growth responses from experiment 1 for **a** juvenile *M. mercenaria* and **b** juvenile *C. virginica*. *Bars* are means ± SE. Slightly negative shell growth for juvenile *M. mercenaria* is within measurement errors of zero. *Letters above bars* indicate significant difference. Nutrients were added as 16:1 inorganic N/P

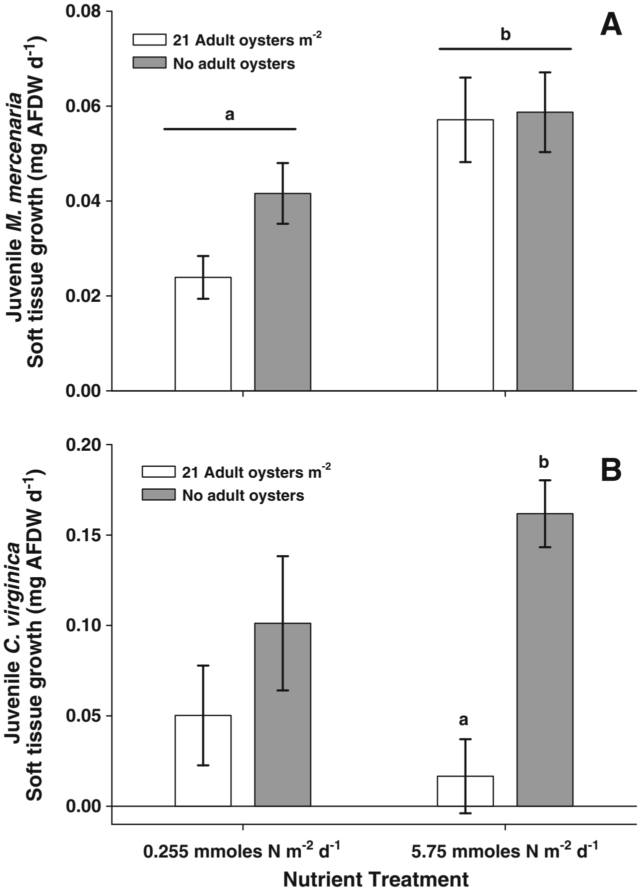

Fig. 4 Growth responses from experiment 2 for **a** juvenile *M. mercenaria* and **b** juvenile *C. virginica*. *Bars* are means ± SE. *Letters above bars* indicate significant difference. Nutrients were added as 16:1 inorganic N/P

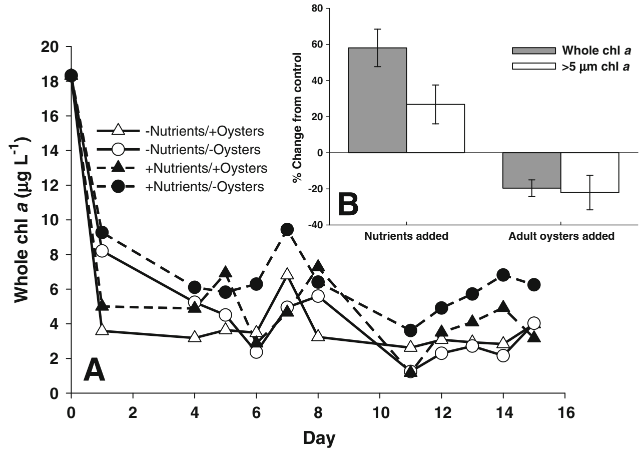

Fig. 3 Chlorophyll *a* dynamics in experiment 2. Time series data points (**a**) represent the mean (*n*=4) for each of the treatment combinations. *Error bars* are not presented for the sake of visual clarity. The mean relative standard deviation of measurements for whole chl *a* was 46.0% during the experiment. *Inset* (**b**) shows mean (± SE) daily percent increase or decrease from nutrient addition over both bivalve treatments and from bivalve addition over both nutrient treatments. See text for magnitudes of nutrient loading and densities of adult bivalves

a in the <5-µm-size fraction varied only by day ($p<0.001$, three-way ANOVAR, data not shown) and was not affected by either treatment ($p>0.05$).

PON was significantly lower in the presence of adult clams (16.1±1.24 µM) compared to the absence of adult clams (38.4±3.37 µM; Table 2; $p<0.05$, two-way ANOVA). POC was affected by both experimental treatments. The levels of POC were higher in the high nutrient loading treatment (249.76±52.82 µM) compared to the low nutrient loading treatment (215.14±35.57 µM; $p<0.05$, two-way ANOVA), and POC was lower in the presence of adult clams (116.88±8.62 µM) compared to the absence of adult clams (310.35±18.92 µM; Table 2; $p<0.001$, two-way ANOVA). The molar ratio of POC/PON was not significantly affected by any of the treatments in experiment 3 (Table 2). Microzooplankton grazing rates were not significantly different between treatments and ranged from 0.45 to 0.73 day^{-1} (Table 2).

Light penetration to the bottom of the mesocosms was higher in the adult bivalve treatment (7,430±437 lux, $p<0.05$, three-way ANOVAR) compared to the absence of adult

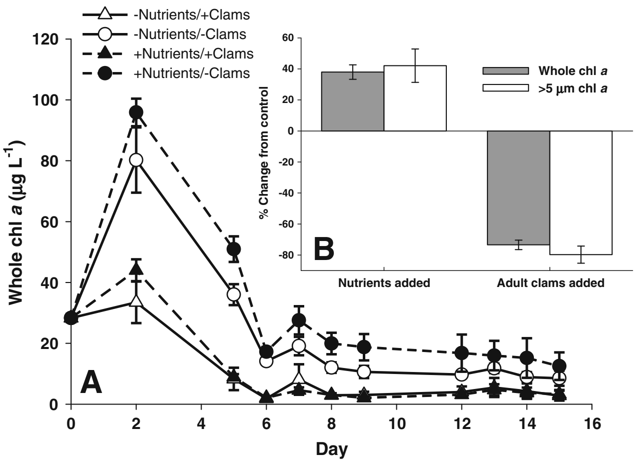

Fig. 5 Chlorophyll *a* dynamics in experiment 3. Time series data points (**a**) represent the mean (± SE, $n=4$) for each of the treatment combinations. *Inset* (**b**) shows mean (± SE) daily percent increase or decrease from nutrient addition over both bivalve treatments and from bivalve addition over both nutrient treatments. See text for magnitudes of nutrient loading and densities of adult bivalves

bivalves (4,620±182 lux), was not significantly affected by the nutrient treatments ($p>0.05$), and varied significantly by day within each mesocosm tank ($p<0.001$, three-way ANOVAR, data not shown). Eelgrass leaf area productivity was significantly enhanced by the presence of adult clams (0.549 ± 0.030 cm^2 $shoot^{-1}$ day^{-1}) compared to the treatments with no adult clams (0.421 ± 0.024 cm^2 $shoot^{-1}$ day^{-1}; Fig. 6a; $p<0.001$, two-way ANOVA). Eelgrass was also affected by the nutrient loading treatment; leaf area productivity was significantly decreased by the high nutrient loading treatment (0.431 ± 0.024 cm^2 $shoot^{-1}$ day^{-1}) compared to the low nutrient loading treatment (0.519 ± 0.029 cm^2 $shoot^{-1}$ day^{-1}; Fig. 6a; $p<0.01$, two-way ANOVA). There was a significant interaction ($p<0.01$) of the treatment effects on eelgrass: The decline in leaf area productivity from the low to the high nutrient loading treatments occurred entirely within the presence of adult clams, while eelgrass growth was not significantly affected by nutrient loading in the absence of adult clams. When eelgrass productivity was measured by dry weight instead of leaf area, there was a significant increase in mass productivity in the presence of adult clams (1.87 ± 0.17 mg $shoot^{-1}$ day^{-1}; Fig. 6a) compared to the absence of adult clams (1.27 ± 0.23 mg $shoot^{-1}$ day^{-1} mass productivity, $p<0.05$, two-way ANOVA). There were no detectable effects of nutrient loading on eelgrass mass productivity. Epiphyte growth on the eelgrass blades was also significantly denser in the presence of adult clams (0.186 ± 0.017 mg AFDW cm^{-2}) compared to the absence of adult clams (0.146 ± 0.009 mg AFDW cm^{-2}; $p<0.01$, two-way ANOVA).

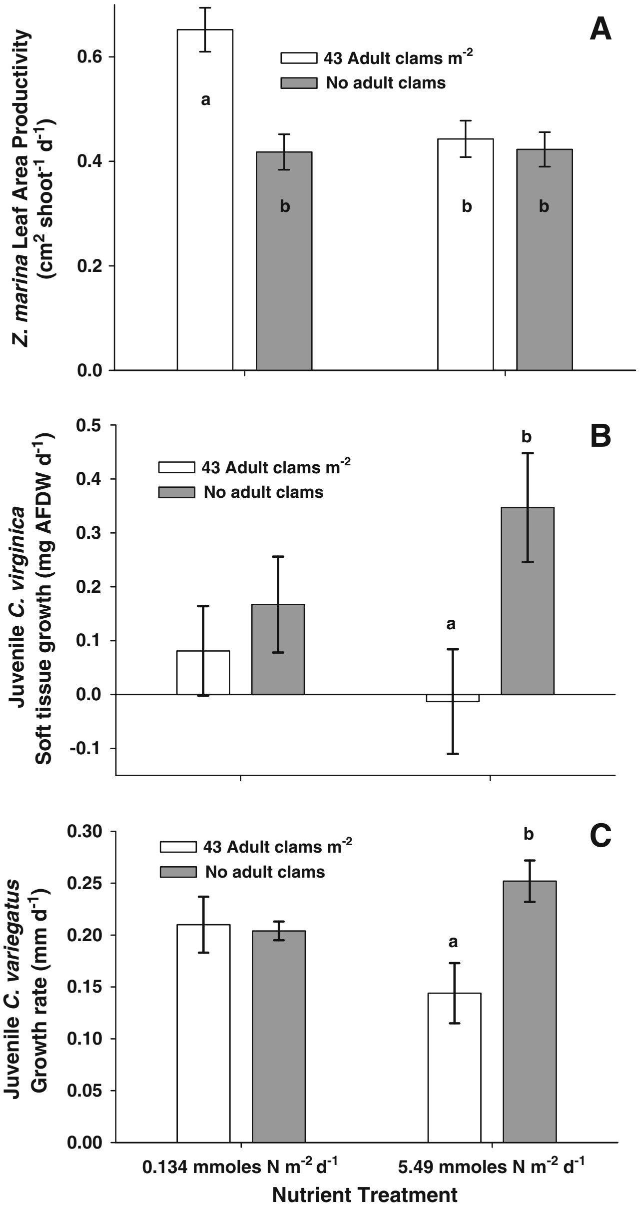

Fig. 6 Growth responses from experiment 3 for **a** *Z. marina*, **b** juvenile *C. virginica*, and **c** juvenile *C. variegatus*. *Bars* are means± SE. *Letters above bars* indicate significant difference. Nutrients were added as 16:1 inorganic N/P

Juvenile clams were not significantly affected by any of the treatment factors in the third experiment. Juvenile oysters grew significantly faster in the absence of adult clams (0.257 ± 0.064 mg AFDW day^{-1}) compared to when adult clams were present (0.034 ± 0.067 mg AFDW day^{-1}; Fig. 6b; $p<0.05$, two-way ANOVA). Juvenile sheepshead minnows also grew significantly faster in the absence of adult clams (0.228 ± 0.017 mm day^{-1}, $p<0.05$, two-way ANOVA) compared to treatments with adult clams (0.177 ± 0.022 mm day^{-1}; Fig. 6c). The fish growth rates showed an interesting interaction: The presence/absence of adult clams made more of a difference to the juvenile sheepshead minnows within the high nutrient loading treatment than within the low nutrient treatment (Fig. 6c; $p<0.05$, Tukey test). There were no differences in juvenile scallop growth rates, but juvenile scallop mortality was significantly higher in the presence of adult clams than in the absence of adult clams (96% with adult clams, 71% without adult clams; $p< 0.001$; *G* test of independence, data not shown). Juvenile fish and shellfish were not significantly affected by the nutrient loading treatments in this experiment (two-way ANOVA).

Discussion

Over the course of three mesocosm experiments, both enhanced nutrient loading and filtration by adult bivalves significantly affected the growth of juvenile shellfish, juvenile fish, and eelgrass, as well as phytoplankton and light levels in mesocosms. The growth of juvenile eastern oysters was the most responsive to the treatment factors; oyster growth was enhanced by high nutrient loading in experiment 1 and decreased by the presence of adult bivalves in experiments 2 and 3. Juvenile northern quahog growth was enhanced by nutrient loading in experiments 1 and 2. The growth of juvenile sheepshead minnows was decreased by the presence of adult bivalves in experiment 3, while the growth of eelgrass shoots was simultaneously increased by adult bivalves and decreased by high nutrient loading in experiment 3. These findings support a conceptual model (Fig. 7), whereby increased nutrient loading acts to increase algal biomass, while increased densities of adult suspension-feeding bivalves decrease phytoplankton abundance. In turn, these changes in phytoplankton biomass affect the growth responses of juvenile bivalves and planktivorous fish through changes in available food particles and affect the growth response of eelgrass through changes in light penetration (Fig. 7). Collectively, these results provide new insight into the manner in which nutrients, and filter feeding bivalves may structure estuarine food webs.

Adult bivalve filtration and nutrient loading were expected to affect eelgrass growth through changes in the density of phytoplankton, which in turn affects the benthic light regime (Newell and Koch 2004; Wall et al. 2008; Fig. 7). Experiment 3 produced results consistent with this hypothesis, where a high density of adult clams decreased chlorophyll *a* levels (Fig. 5a, b) and increased light penetration leading to an increase in eelgrass productivity (Fig. 6a). The high nutrient loading treatment in experiment 3 decreased eelgrass growth relative to the low nutrient loading treatment, and the effects of nutrient loading on eelgrass were most evident when adult clams were present (Fig. 6a). Adult clams and adult oysters decreased chlorophyll *a* in experiments 1 and 2 similarly to experiment 3 (Figs. 1a, b and 3a, b), but these changes in chl *a* did not produce significant effects on light or eelgrass. The density of epiphytes on eelgrass blades was increased by adult bivalve filtration in experiments 2 and 3 and by nutrient loading in experiment 2. Although thick epiphyte growth has been found to have a negative impact on seagrass in some cases (Duarte 1995), the densities of epiphytes measured in our experiments (0.13–0.19 mg AFDW cm^{-2}) were likely too low to significantly block light at the blade surface (Brush and Nixon 2002). On a time-scale longer than these experiments (weeks–months), nutrient enrichment (or lack thereof) to the sediments will also affect seagrass growth and reproduction (Dennison et al. 1987; Peterson and Heck 2001; Carroll et al. 2008).

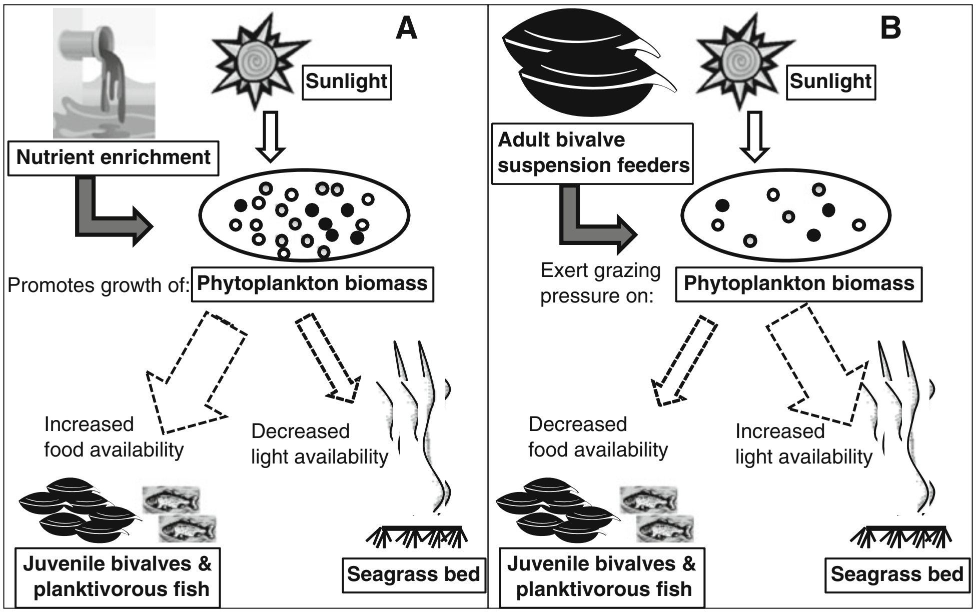

Fig. 7 Conceptual model of the effects of nutrient loading (**a**) and adult bivalve filtration (**b**) in mesocosm experiments. *Solid gray arrows* represent treatment factors that are expected to act on phytoplankton biomass, and *dashed arrows* represent hypothesized responses of juvenile bivalves, juvenile planktivorous fish, and seagrass shoots to changes in phytoplankton biomass

Growth rates of juvenile planktivorous fish and juvenile bivalves may be decreased by filtration pressure from adult bivalves, which clear food particles from the water column (Rheault and Rice 1996; Zhou et al. 2006), or may be increased by high nutrient loading, which may increase the quantity and quality of suspended food particles (Carmichael et al. 2004; Carmichael and Valiela 2005; Fig. 7). All three experiments had some results consistent with this hypothesis: Juvenile clam growth was increased by high nutrient loading in experiments 1 and 2 (Figs. 2a and 4a), juvenile oyster growth was also increased by high nutrient loading in experiment 1 (Fig. 2b), while juvenile oyster growth was decreased by adult bivalve filtration in experiments 2 and 3 (Figs. 4b and 6b), and juvenile fish growth was also decreased by adult bivalve filtration in experiment 3 (Fig. 6c). Although there were no significant growth responses for scallops, juvenile scallop mortality was increased by adult bivalve filtration in experiment 3.

The results of these experiments demonstrate the strong reliance of juvenile shellfish and planktivorous finfish growth rates and survival on the short-term dynamics (days to weeks) of food availability as reflected by concentrations of chlorophyll *a*, POC, and PON (Fig. 7). In experiment 1, where nutrient loading had a strong effect on juvenile growth, the molar ratio of POC/PON was significantly reduced by the high nutrient loading treatment (Table 2), suggesting an enrichment of nitrogen in food particles could have contributed to enhanced shellfish growth (Fig. 2a, b). Carmichael et al. (2004) and Carmichael and Valiela (2005) have interpreted nitrogen-enriched seston as an increase in the quality of food particles available to juvenile bivalves. Although the molar ratio of POC/PON did not change in experiment 3, the quantities of POC and PON were both decreased by adult clam filtration (Table 2), with corresponding decreases in the growth rates of juvenile oysters and sheepshead minnows (Fig. 6b, c) and a decrease in the survival of juvenile scallops. In all cases, increased growth rates of shellfish occurred in parallel with increases in whole or size-fractionated chlorophyll *a*. While there was a statistically significant change in chl *a* due to treatment factors in each experiment, the magnitude of these changes in experiments 1 and 2 were relatively low (±2–4 μg L^{-1}; Figs. 1a and 3a). It is possible that the availability of food particles to the juvenile shellfish was changed by the treatment factors in these experiments without large changes in the standing stock of chlorophyll *a* between treatments. Phytoplankton mortality rates due to microzooplankton grazing of 0.5 day^{-1} or greater are common in estuarine environments and often result in >70% daily turnover of standing chl *a* (Calbet and Landry 2004). Microzooplankton grazing rates ranged from 2.3 to 2.4 day^{-1} in experiment 2; these were faster than the estimated clearance rate from adult oyster filtration of 67% tank volume day^{-1}. Such rapid rates of phytoplankton community turnover could mask true food availability to juvenile bivalves and would account for enhanced bivalve growth responses in experiment 2 in the absence of large changes in chl *a*. In experiment 3, microzooplankton grazing rates were slower (0.4–0.7 day^{-1}) and comparable to the adult clam clearance rate of 63% tank volume day^{-1}. In contrast to experiments 1 and 2, this experiment had large treatment-driven changes in chl *a* (±20–40 μg L^{-1}; Fig. 5a) and growth differences in response to adult clam filtration (Fig. 6a–c). Lonsdale et al. (2009) found that natural populations of bivalves in a shallow embayment could exert grazing pressure on phytoplankton that was comparable to grazing by microzooplankton and noted that bivalves also fed upon microzooplankton and copepod nauplii. Future work will need to examine the extent to which benthic suspension feeding alters both phytoplankton growth and microzooplankton grazing and how turnover in the plankton community affects the growth and recruitment of juvenile bivalves.

During these experiments, the treatment factor driving the growth responses changed from nutrient loading in the first experiment to combined factors in the second experiment and finally to exclusively adult bivalve filtration in the third experiment. These differences may partly reflect differences in treatment administered: Experiment 1 had a larger difference in nutrient loading rate between the high nutrient treatment and the control than the other experiments, while experiment 3 had a larger difference in clam density between adult clam treatments than experiment 1 (Table 1). These results may have also been influenced by seasonal trends: Lower temperatures during the first experiment (17–23°C) may have yielded lower nutrient regeneration rates (Nagata and Kirchman 1992; Miller et al. 1995) and low bivalve filtration rates (Kraueter and Castagna 2001), making external nutrient loading a more important process. Conversely, higher temperatures (23–25°C) for the second and third experiments likely promoted faster bivalve filtration (Kraueter and Castagna 2001) and pelagic nutrient regeneration (Nagata and Kirchman 1992; Miller et al. 1995). There is also evidence of seasonal succession in the phytoplankton community, since the <5-μm-size fraction of chl *a* responded more strongly to the treatment factors in experiments 1 and 2 (June and July) while the >5-μm-size fraction responded more strongly in experiment 3 (Aug). As such, it seems that bivalve filtration can mediate the eutrophication of estuarine food webs, and the relative importance of this mediating role can change seasonally or with changing rates of nutrient loading or densities of bivalves.

The densities of adult northern quahogs used in our experiments 1 and 3 (8–12 individuals $tank^{-1}$, or 29–43 individuals m^{-2}) are comparable to historic densities of northern quahogs (hard clams) in Great South Bay (50–100 individuals m^{-2}, Kassner 1993, cf. Cerrato et al. 2004) but are much higher than current densities in NY estuaries (0–5 individuals m^{-2}, Weiss et al. 2007). Similarly, the density of adult oysters used in experiment 2 (6 individuals $tank^{-1}$, or 21 individuals m^{-2}) is comparable to historic densities of Eastern oysters in reefs in Chesapeake Bay (43–150 individuals m^{-2}) but is much higher than current densities (0.43 individuals m^{-2}; Newell 1988; MacKenzie 1996). However, all of the densities used in experiments are several orders of magnitude less than levels used for bivalve aquaculture (Rheault and Rice 1996; K. Rivara, Aeros Cultured Oyster Co., personal communication). The estimated water column clearance rates from these densities of adult bivalves were 42–67% tank volume day^{-1}, within the range reported to control algal bloom formation (Cerrato et al. 2004; Wall et al. 2008). Consistent with this idea, the presence of adult bivalves yielded lower phytoplankton biomass in all three experiments (Figs. 1b, 3b, and 5b). Such ecosystem-wide filtration pressure may have been typical of historic (nineteenth century) natural bivalve populations in Chesapeake Bay (Newell 1988; MacKenzie 1996) or Great South Bay (mid-twentieth century, McHugh 1991; Kassner 1993). Similarly, modern high-density bivalve aquaculture may also achieve these ecosystem filtration rates (Dumbauld et al. 2009), especially in coastal lagoons with slow flushing times (Souchu et al. 2001) and in some cases the loss of filtration due to the removal of bivalve aquaculture can lead to symptoms of eutrophication (Huang et al. 2008). Estuarine management programs may consider bivalve restoration as a management tool to control pelagic algal blooms (Cerrato et al. 2004), combat eutrophication (Cerco and Noel 2007), facilitate the growth of eelgrass (Fig. 6a; Peterson and Heck 2001; Newell and Koch 2004; Wall et al. 2008), or even to effect “regime change” of eutrophic estuaries (Petersen et al. 2008), although the potential impacts on juvenile shellfish must also be considered.

While enhanced bivalve filtration was beneficial to eelgrass and to some extent epiphytes on eelgrass, they exerted a significantly negative effect on the growth of juvenile fish and shellfish in two out of three experiments (Figs. 4b and 6b, c) and in one case even led to a significant increase in juvenile scallop mortality (experiment 3). Rheault and Rice (1996) placed juvenile eastern oysters (*C. virginica*) and bay scallops (*A. irradians*) in a compartmented flume and found decreased growth and condition index in the shellfish that were downstream compared to the upstream dense populations. In experiment 3 of our study, the high density of adult clams produced a large average daily drop in chl *a* levels (Fig. 5b, −73%) and a decrease of 36% in experiment-long chl *a* means compared to the control and also led to decreased growth of juvenile oysters (Fig. 6b) and decreased survival of juvenile scallops. The concentrations of chl *a* in experiment 3 were relatively high (25.09±2.56 μg L^{-1} with no adult clams; 15.90±2.20 μg L^{-1} with adult clams; Table 2); this drop in chlorophyll *a* produced a significant decrease in juvenile oyster growth but not juvenile clam growth. It is likely that juvenile clam food requirements were saturated at a lower chlorophyll *a* concentration than juvenile oyster food requirements (Tenore and Dunstan 1973). These impacts illustrate an eventual trade-off between the benefits and costs of higher ecosystem filtration rates: Despite the benefits to seagrass, high rates of water column turnover by adult shellfish could serve as a negative feedback on juvenile fish and shellfish populations (Figs. 4b and 6b, c) by decreasing food availability (Fig. 5a, b) or even by direct consumption of larval bivalves by adults (Andre and Rosenberg 1991; Andre et al. 1993). Such density-dependent limitation is a common phenomenon within bivalve aquaculture (Rheault and Rice 1996; Zhou et al. 2006), and overstocking of aquaculture operations may exceed the carrying capacity of some estuaries (Guo et al. 1999; Duarte et al. 2003). The extent to which juvenile suspension feeders may be food-limited within estuarine ecosystems is not well-known but will certainly depend on the species involved and the particular physics and biology of each ecosystem (Newell 2004; Ferreira et al. 2008).

Many estuarine management plans have focused on the need to reduce nutrient loads to mitigate the effects of eutrophication (Nixon 1995; Cloern 2001; de Jonge et al. 2002). Partly through changes in land use and better sewage treatment, inorganic nutrient levels and/or chlorophyll *a* concentrations have declined in some coastal waters, such as the North Sea (Nunneri et al. 2007; Artioli et al. 2008), the Dutch Wadden Sea (Philippart et al. 2007), Narragansett Bay, RI, USA (Fulweiler et al. 2007), Long Island Sound, USA (CTDEP 1991–2007), and the Peconic Estuary, NY, USA (SCDHS 1976–2005). Despite this “oligotrophication” of some coastal waters (Nixon et al. 2009), the recovery of estuarine resources in these systems has not been reported. The high nutrient loading rates in our experimental tanks are comparable to measured nutrient loading rates in eutrophic northeast US estuaries (Taylor et al. 1999), from which valuable estuarine resources have been lost (Ryther 1989; McHugh 1991; Valiela et al. 1992). However, positive effects on bivalves under enhanced levels of nutrient loading have been reported (Reitan et al. 2002; Weiss et al. 2002; Carmichael et al. 2004). Eutrophic systems with high levels of nutrient loading often have hypoxia/anoxia (Nixon 1995; Diaz and Rosenberg 2008), which can decrease bivalve survival (Carmichael et al. 2004), but our well-mixed mesocosms remained normoxic (>4 mg L^{-1} DO) during experiments. Considering this information, our findings suggest that nutrient loading could be allowed to increase in some relatively oligo- or mesotrophic and well-mixed coastal systems with increased secondary production of eastern oysters and northern quahogs as a positive benefit (Nixon and Buckley 2002). Of course, such potential benefits would need to be considered in light of potentially negative effects of higher nutrient loads in an ecosystem such as hypoxia (Diaz and Rosenberg 2008), loss of seagrass beds (Valiela et al. 1992; Dennison et al. 1989), and harmful algal blooms (Anderson et al. 2008).

Future ecosystem-based management of estuaries will need to simultaneously administer bivalve restoration, control of nutrient loading, conservation of key fishery species, the burgeoning aquaculture industry, and protection of critical habitats such as seagrass meadows and salt marshes. Quantitative modeling of bivalve filtration, phytoplankton dynamics, and hydrology of estuaries will aid in the aforementioned management goals (Dame and Prins 1998; Duarte et al. 2003; Ferreira et al. 2008). Based on the results of these experiments and other findings, some general conclusions can be drawn. First, eelgrass is light-limited in many eutrophic estuaries (Dennison and Alberte 1985; Duarte 1995) and will benefit from proximity to the enhanced filtration of bivalve beds (Fig. 7; Wall et al. 2008). Additionally, bivalves can benefit seagrasses through enhanced biodeposition (Peterson and Heck 2001; Carroll et al. 2008). As such, re-planting of eelgrass beds should focus on areas that have high light penetration and/or are adjacent to existing dense bivalve populations. The second conclusion is that juvenile resource bivalves can respond positively to enhanced nutrient loading but may experience decreased growth in the presence of high densities of adult bivalves (Fig. 7). This is likely mediated by food limitation: Nutrients encourage the growth of larger and more nutritious phytoplankton (Wikfors et al. 1992; Raven and Kubler 2002) while dense collections of adult bivalves can limit juvenile growth by clearing too many of these food particles (Rheault and Rice 1996; Zhou et al. 2006). This issue of food limitation between juvenile and adult bivalves may be best seen through the lens of intensifying aquaculture operations: As aquaculture becomes more prevalent and shellfish stocking densities increase, aquaculture operations may limit each other or adjacent natural populations (Nunes et al. 2003; Ferreira et al. 2008). Clearly, predators (Gosselin and Qian 1997; Polyakov et al. 2007) and hypoxia/anoxia (Altieri and Witman 2006; Diaz and Rosenberg 2008) also exert significant mortalities on juvenile bivalves in the estuarine ecosystems. However, in absence of hypoxia and differential predation, restoration, re-seeding, and aquaculture of bivalves are more likely to succeed in areas that have moderate nutrient loading rates, although managers must carefully consider the spacing between both natural and aquacultured bivalve populations.

Acknowledgments Many people provided field and laboratory support for this project including A. Burson, B. Rodgers, J. Goleski, M. Marcoval, A. Stubler, C. Blankenship, J. Carroll, C. Thomas, S. Brisbin, T. Duffy, T. Davis, L. Davis, F. Koch, J. Myers, S. Talmage, T. Hattenrath, T. Behling, D. Getz, M. Meade, B. Gagliardi, and M. Harke. We thank Dr. Evan Ward for guidance in performing clearance rate measurements. We thank Drs. Evan Ward, Glenn Lopez, and Robert Aller for useful comments on a prior version of this manuscript. Juvenile shellfish were provided by G. Rivara and M. Patricio at Cornell Cooperative Extension, Southold, NY, USA. This project was supported by The Nature Conservancy and the New Tamarind Foundation.

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Helgol Mar Res (2011) 65:275–283
DOI 10.1007/s10152-010-0221-4

ORIGINAL ARTICLE

Population structure of the ribbed mussel *Geukensia demissa* in salt marshes in the southern Gulf of St. Lawrence, Canada

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Received: 6 October 2009 / Revised: 5 August 2010 / Accepted: 6 August 2010 / Published online: 22 August 2010

Abstract The ribbed mussel, *Geukensia demissa,* is highly dependent on the cordgrass *Spartina alterniflora* for amelioration from environmental stress and substrate stabilization. *Spartina alterniflora* is a foundation species in marshes, and *G. demissa* is typically associated with cordgrass beds. Marshes in the southern Gulf of St. Lawrence are experiencing erosion and degradation, presumably as a result of increases in sea level, which increases salinity exposure and negatively impacts *S. alterniflora*. The population structure of the ribbed mussel, *Geukensia demissa*, was studied at nine sites in six estuaries in the southern Gulf of St. Lawrence in Nova Scotia, Canada, where marsh degradation is occurring. Mussel length was used as a proxy for age of *G. demissa* in three salt marsh zones characterized by density and elevation of *Spartina alterniflora*: (1) a lower zone in which the *S. alterniflora* was dead, but where the basal mat was coherent, (2) a zone of living, but low density *S. alterniflora* at the margin of the living marsh, and (3) a zone of dense *S. alterniflora* one to three meters back from the edge. Mussel length was significantly different across the three zones in seven of the nine sites. Mean length decreased as elevation increased, and small mussels (i.e., 1–3 cm) were absent at seven sites. The smallest mussels occurred in the dense *S. alterniflora* zone, higher in the marsh. Mussel length in the two western sites did not differ between zones, and small mussels (i.e., 1–3 cm) were present, but rare. The absence of small mussels in seven of the nine sites, and the size frequency distribution at remaining sites, suggests a lack of recent recruitment and a long-term threat to the survival of *G. demissa.* Salt marsh degradation and the death of *S. alterniflora* have negatively impacted *G. demissa* recruitment, and population decline is evident.

Keywords Ecosystem engineers · *Geukensia demissa* · Population structure · Salt marsh erosion · Sea level rise · *Spartina alterniflora*

Introduction

Ecosystem engineers are organisms that modify, maintain or create habitat by impacting the availability of resources through physical changes in biotic or abiotic materials (Jones et al. 1994). Autogenic engineers change the environment through their physical structure, while allogenic engineers transform biotic or abiotic material (Jones et al. 1994). These organisms are often the focus of conservation efforts because they are known to positively impact species richness and diversity (Castilla et al. 2004; Bangert and Slobodchikoff 2006; Borthagaray and Carranza 2007; Buse et al. 2008; Bouma et al. 2009; Bravo et al. 2009), and they have cascading impacts on communities (Coleman and Williams 2002). Salt marshes along the northwestern Atlantic exhibit strong zonation of flowering plants. At the lowest marsh elevations, *Spartina alterniflora* Loisel. dominates, while at higher elevations, other flowering plants including *S. patens* (Ait.) Muhl., *Juncus* spp. and *S. pectinata* Link become dominant and successively exclude species beginning with *S. alterniflora* (Bertness 1991; Davis and Browne 1997). These marshes contain two ecosystem engineers that work together to stabilize the marsh substrate. *Spartina alterniflora* is considered as an

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Communicated by H.-D. Franke.

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autogenic ecosystem engineer in marsh ecosystems because it modifies environmental conditions by aerating anoxic sediments (Arenovski and Howes 1992) and reducing flow velocity and water turbidity (Leonard and Luther 1995; Leonard and Croft 2006), as well as stabilizing the marsh substrate and providing attachment sites for other organisms (Altieri et al. 2007). *Geukensia demissa* (Dillwyn) is a marsh mussel found half embedded in the mud among the rhizomes and roots of *S. alterniflora* (e.g., Bertness 1984; Jost and Helmuth 2007). These mussels are both autogenic and allogenic ecosystem engineers as they enhance nutrient availability for *S. alterniflora*, bind and protect sediments and inhibit marsh erosion and degradation (Bertness 1984; Jones et al. 1994). These two ecosystem engineers work together in marshes to increase species richness and abundance through a facilitative cascade where *S. alterniflora* is the foundation ecosystem engineer and *G. demissa* is a secondary engineer that further enhances the impacts of *S. alterniflora* (Altieri et al. 2007). *Geukensia demissa* abundance and byssal thread attachment strength is significantly reduced when *S. alterniflora* is removed from the marsh (Altieri et al. 2007). Individual and population growth of *G. demissa* decreases with decreased *S. alterniflora* production (Stiven and Kuenzler 1979). Altieri et al. (2007) found *S. alterniflora* was able to inhabit new areas and successfully establish without the presence of *G. demissa*; however, *G. demissa* did not flourish without the environmental amelioration provided by *S. alterniflora* through shading and stabilizing the substrate. Furthermore, the mussels are only found in cordgrass beds in marshes in New England; thus, there appears to be a high dependence of *G. demissa* on *S. alterniflora* in this area (Altieri et al. 2007).

Despite the importance of these species in maintaining marshes, no study has investigated their relationship in northern marshes. The vast majority of studies on interactions between *S. alterniflora* and *G. demissa* have been conducted in New England (Bertness 1984; Altieri et al. 2007); however, these marshes do not experience the harsh environmental conditions and increased storm frequency that occurs at latitudes further north (Wimmer et al. 2006; Bertness 2007). Marshes along the southern coast of the Gulf of St. Lawrence, Canada are known to be deteriorating. In healthy marshes, there are clear zones due to the distribution of flowering plants at various elevations; however, in the southern Gulf of St. Lawrence, the marsh edge is fringed with a mat of peat where *S. alterniflora* had once flourished, but has subsequently died off, and protruding from these dead mats are *G. demissa* (Garbary unpublished). The southern Gulf of St. Lawrence experiences greater environmental stress than New England marshes; therefore, we predicted *G. demissa* would exhibit an even greater dependence on *S. alterniflora* and that recruitment of *G. demissa* to the lowest marsh elevations would be inhibited by the lack of living *S. alterniflora*. Our study set out to investigate the general patterns of *G. demissa* distribution across zones of *S. alterniflora* in the deteriorating marshes along the southern coast of the Gulf of St. Lawrence, Canada. We examined the size structure of populations of *G. demissa* across zones of *S. alterniflora* (defined based on the density of *S. alterniflora*).

Materials and methods

Size structure of *Geukensia demissa* was examined at nine sites along the north shore of Nova Scotia, on the Gulf of St. Lawrence in the late summer and fall of 2007 and 2008. Two sites were located in Pomquet, Merigomish and Pugwash Harbours and one site in Cheticamp, Antigonish and Caribou Harbours (Fig. 1, Table 1). All of these harbors are estuaries formed from drowned river valleys (Davis and Browne 1997). Cheticamp, Pomquet, Merigomish, Antigonish and Caribou Harbour are all shallow water systems (mostly 1–5 m) with sand dune barrier beaches and with soft-bottom communities typically dominated by *Zostera marina* L. Pugwash Harbour is a deeper water channel with no sand dune barrier present. All the estuaries have extensive areas of fringing salt marshes and a tidal amplitude of approximately 1.4 m. The shores have very limited exposure to wave action, and the fetch for the sampling sites varied from 200 to 500 m. Although there is considerable variation in salinity within each estuary, sampling sites we used typically varied from 20 to 28‰ (Kim et al. 2004; Garbary et al. 2008).

Within the region of the marsh dominated by *S. alterniflora*, we defined three zones based on its density and elevation (Fig. 2). The first zone consisted of mats of peat

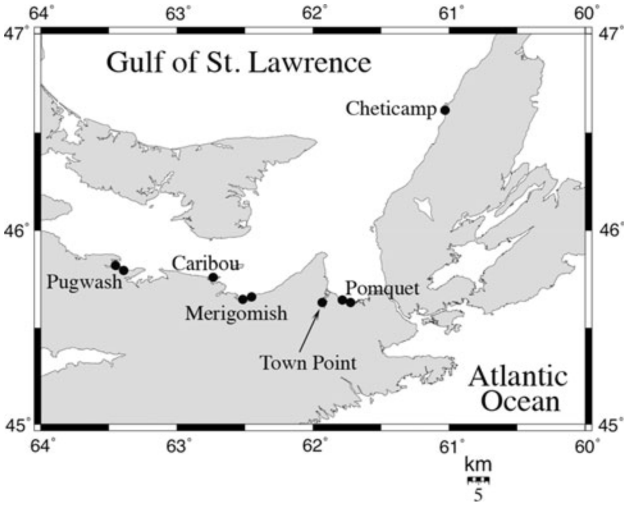

Fig. 1 Map indicating all sites sampled in Nova Scotia, Canada

Table 1 Estuary, study site with coordinates and the years each study site was sampled or visited

Estuary	Study site	Longitude/latitude	Dates sampled/visited
Cheticamp Harbour	Cheticamp	61°02′1″/46°36′2″	August 2008/2009
Pomquet Harbour	Pomquet 1	60°50′2″/45°38′9″	September 2007/visited 2008/2009
	Pomquet 2	60°49′4″/45°38′7″	September 2007/visited 2008/2009
Antigonish Harbour	Town Point	62°54′0″/45°40′3″	September 2007/2008/2009
Merigomish Harbour	Merigomish 1	62°26′3″/45°37′6″	October 2007
	Merigomish 2	62°25′1″/45°38′1″	October 2007
Caribou Harbour	Caribou	62°39′5″/45°44′2″	October 2007
Pugwash Harbour	Pugwash 1	63°39′0″/45°50′6″	August 2008
	Pugwash 2	63°39′5″/45°50′4″	August 2008

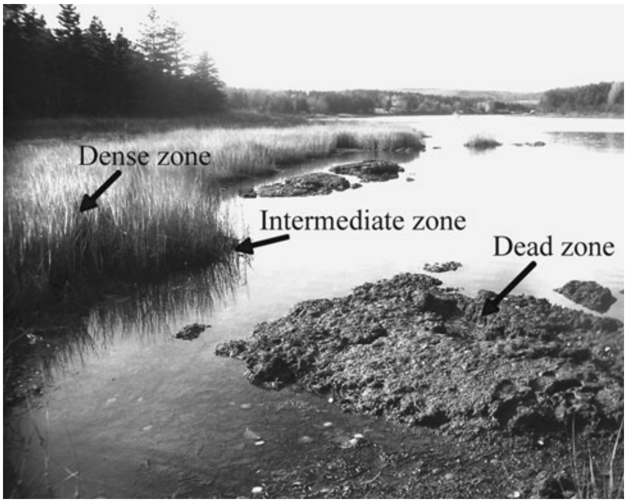

Fig. 2 Portion of salt marsh at Town Point, Antigonish Harbour with arrows indicating the three *Spartina alterniflora* zones

where *S. alterniflora* had once grown, but had died and was at the lowest marsh elevations. At some sites, the dead zone was contiguous with the living marsh vegetation; at other sites, the dead mats had become detached and drifted to slightly lower intertidal elevations. The width of this zone ranged from 0.6 to 4.0 m (average ca. 1.7 m), as measured from the widest area where mussels were sampled. At zone margins, the zones were sometimes indistinct and, therefore, *G. demissa* was defined as being in the dead zone if there was no living *S. alterniflora* within 5.0 cm around the mussel. The 'living' marsh vegetation begins with a region in which *S. alterniflora* appears healthy, but occurs in relatively low density and was higher in elevation than the dead zone. We refer to this as the intermediate zone. The zone was adjacent to the dead zone but slightly higher in elevation (ca. 10 cm). Within this zone, there was living *S. alterniflora*; however, it was sparse. The width of this zone across the sites was 1.0–2.8 m (average ca. 1.7 m). *Geukensia demissa* was defined as being a part of the intermediate zone if it had at least one living *S. alterniflora* shoot within 5.0 cm. The final zone was referred to as the dense *S. alterniflora* zone, and it was between the intermediate zone and a region of marsh dominated either by *Spartina patens* or by *Juncus gerardii* and was at the highest marsh elevation considered. *Geukensia demissa* in the dense zone was surrounded by *S. alterniflora* on all sides and was often attached to the underground root-rhizome system by byssal threads. The width of this zone across the sites was 1.2–6.4 m (average ca. 2.9 m).

The lengths of 150 *G. demissa* were measured per site, 50 from each of the dead, intermediate and dense zones. *Geukensia demissa* population numbers were low in all sites, and the first 50 mussels encountered during an intensive search, beginning at one edge of the marsh, were measured. The mussels were measured from the anterior to the posterior ends. Although counting internal growth rings provides an accurate measure of age, we chose length as an approximation of age because this method is non-destructive. In addition, previous studies found that length of *G. demissa* is highly correlated with age (Lutz and Castagna 1980; Brousseau 1984; Bertness and Grosholz 1985). Analysis of external growth bands has also been used to age mussels; however, this procedure is not as accurate because annual rings are not always clearly distinguished from other concentric lines and scars on the shell surface (Brousseau 1984). These rings can be hard to distinguish once the mussel ages and shell erosion occurs. Furthermore, we considered this method too intrusive.

The density of mussels within each zone was determined using a 50 × 50 cm quadrat (0.25 m^2) (Fig. 3). Quadrats were placed on both sides of a five meter transect line within each zone. The number of mussels within the quadrat was counted ($n = 10$ for each zone at each site). Density measurements were only conducted at three sites because the population of *G. demissa* was too small at the other sites (<4 mussels m^{-2}). However, we do not believe that densities at any of the sites were large enough to cause density-driven effects on recruitment. Mussel density was analyzed in a two-way ANOVA with Site (random; 3

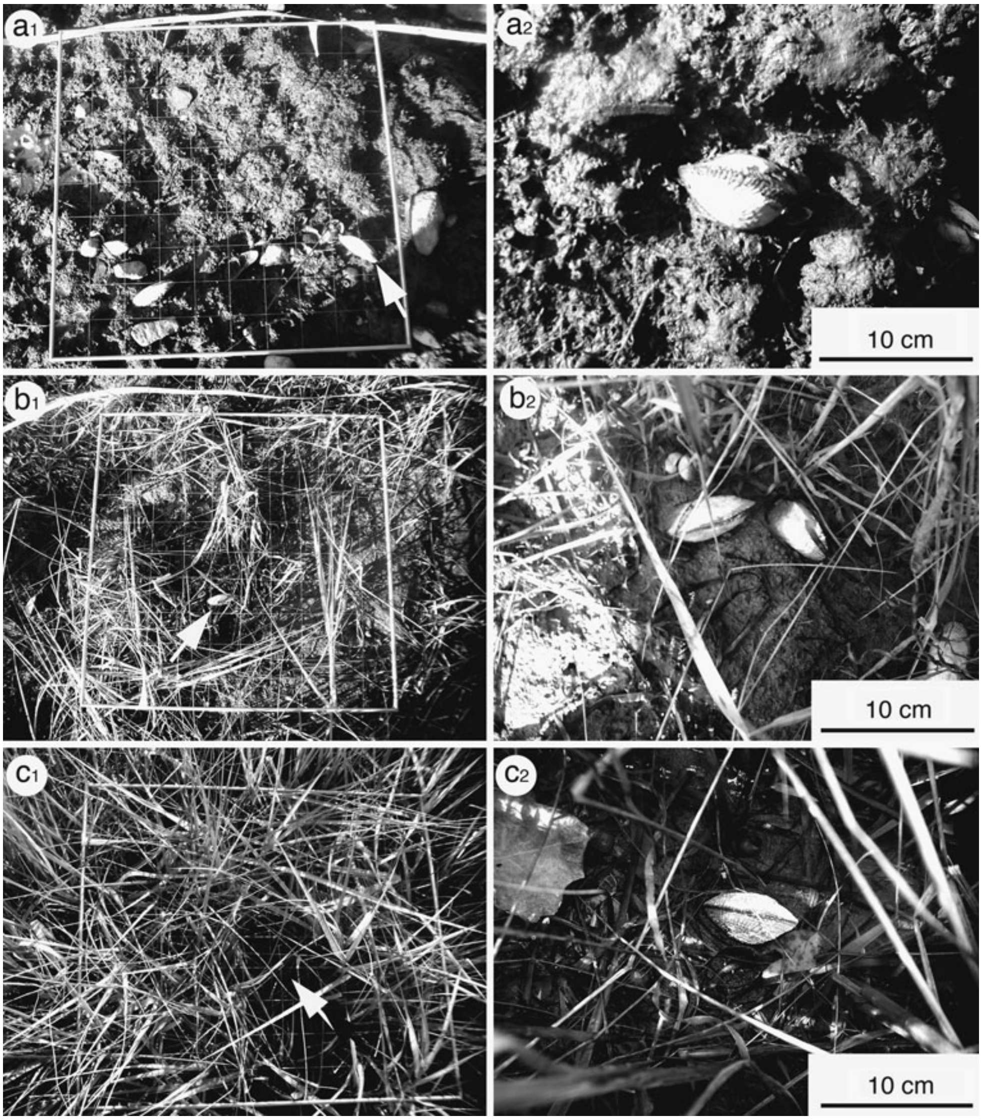

Fig. 3 Photograph of a quadrat on each of the three *Spartina alterniflora* zones; adjacent photograph indicates *Geukensia demissa* in each of the zones. *Arrows* indicate the mussel within the quadrat that is blown up in $\mathbf{a}_2$, $\mathbf{b}_2$ and $\mathbf{c}_2$. $\mathbf{a}_1$, $\mathbf{a}_2$ Dead zone, $\mathbf{b}_1$, $\mathbf{b}_2$ intermediate zone and $\mathbf{c}_1$, $\mathbf{c}_2$ dense zone

levels), Zone (fixed; 3 levels) and the interaction between the two factors (Site * Zone). Tukey's tests were used to determine where these differences occurred. All statistical tests were conducted using JMP for Mac.

Three sediment samples of 20 × 20 cm and 10 cm deep were taken in each zone at Town Point to determine whether there were any small mussels buried in the sediment that would have been missed during sampling. The samples were cut out of the sediment and brought back to the laboratory to be washed through a 0.5-cm mesh sieve.

Mussel length was analyzed in a two-way ANOVA with Site (random; 9 levels) and Zone (fixed; 3 levels) and the interaction between the two factors (Site * Zone). When a significant interaction was detected, simple effects were investigated utilizing one-way ANOVAs at each level of the other factor (Underwood 1997). For instance, differences among zones were investigated at each site, and significant differences among sites were investigated within each zone (Underwood 1997). When significance was obtained in the ANOVA, multiple comparisons were made using Tukey's honestly significant difference tests. Normality was assessed using normal quantile plots, and homogeneity of variance was tested across all treatment combinations simultaneously using Levene's test.

We returned to the sites in Pomquet and Antigonish Harbours in 2008 and 2009 and to the Cheticamp site in 2009 to observe the *G. demissa* populations. Fifty *G. demissa* in the intermediate and dense zones were measured at Town Point in 2008 and 2009. There were fewer than 50 mussels in the dead zone; thus, rather than disturbing the already small population, the mussels were not measured, but the total number of mussels was recorded. At the Cheticamp site, it was easy to locate 50 mussels in all zones in 2008; however, upon return to the site in 2009 after extensive searching, there were only 15 mussels located in the dense zone. Thus, analyses comparing mussel length at Cheticamp over time were conducted only on the dead and intermediate zones. Populations of *G. demissa* had reduced to the point that it was no longer possible to locate 50 mussels in any of the zones at the sites in Pomquet; thus, no measurements were conducted.

To determine whether mussel length varied over time, length data at Town Point in 2007, 2008 and 2009 were analyzed in a two-way ANOVA with Time (fixed; 3 levels) and Zone (fixed; 2 levels) and the interaction between the two factors (Time * Zone); mussel length at Cheticamp was analyzed in the same fashion but Time had only 2 levels (2008 and 2009). When a significant interaction was detected, simple effects were investigated for each Time utilizing *t*-tests, and ANOVA was utilized for each Zone to determine whether there was a significant difference between sampling times. Normality was assessed using normal quantile plots, and homogeneity of variance was tested across all treatment combinations simultaneously using Levene's test.

Results

Mussel length data were normally distributed, but failed to meet the homogeneity of variance assumption (Levene's test, $F_{26,\ 1323} = 5.7597$, $P < 0.05$), and no transformation resulted in homogenous variances; however, when conducting experiments with multiple treatments and large sample sizes (>6 in each treatment) the ANOVA is considered robust (Box 1953; Underwood 1997). There was a significant difference among sites (ANOVA, $F_{8,\ 1323} = 122.8187$, $P < 0.0001$) and zones (ANOVA, $F_{2,\ 1323} = 65.5873$, $P < 0.0001$), and a significant interaction between site and zone (ANOVA, $F_{16,\ 1323} = 6.0863$, $P < 0.0001$). Simple effects on each site revealed that there was no significant difference in mussel length among zones in Cheticamp (ANOVA, $F_{2,\ 147} = 2.1682$, $P = 0.1146$) or Pugwash 2 (ANOVA, $F_{2,\ 147} = 2.1022$, $P = 0.1258$); however, in all other sites, there were significant differences among zones ($P < 0.05$) (Fig. 4). Mussel length was greatest in the dead zone and least in the dense zone in all sites where differences occurred (Tukey's test, $P < 0.05$), except in Pugwash 2 where mussel length was greatest in the dense zone and least in the intermediate zone (Tukey's test, $P < 0.05$) (Fig. 4). Simple effects on each zone revealed there were significant differences among sites in the dead (ANOVA, $F_{8,\ 441} = 67.1825$, $P < 0.0001$), intermediate (ANOVA, $F_{8,\ 441} = 51.7704$, $P < 0.0001$) and dense (ANOVA, $F_{8,\ 441} = 21.4245$, $P < 0.0001$) *S. alterniflora* zones. In the dead zone, Town Point had the largest mussels, and sites in Pugwash Harbour had the smallest (Tukey's test, $P < 0.05$) (Fig. 4). Mussel length in the intermediate zone was greatest at Pomquet 2 and least at the sites in Pugwash Harbour (Tukey's test, $P < 0.05$) (Fig. 4). In the dense zone, mussel length was greatest at Town Point and the sites in Pomquet Harbour and least in

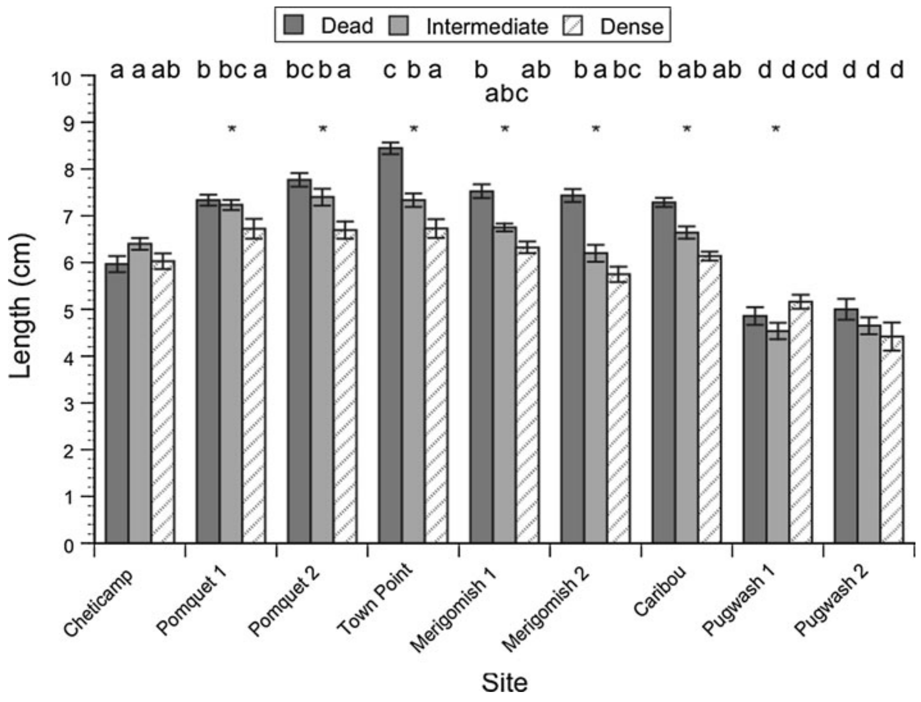

Fig. 4 Mussel length in the three *Spartina alterniflora* zones at each site. *Asterisks* indicate significant differences among zones within each site, while different letters indicate significant differences among sites within each zone. *Error bars* represent standard error; $n = 50$

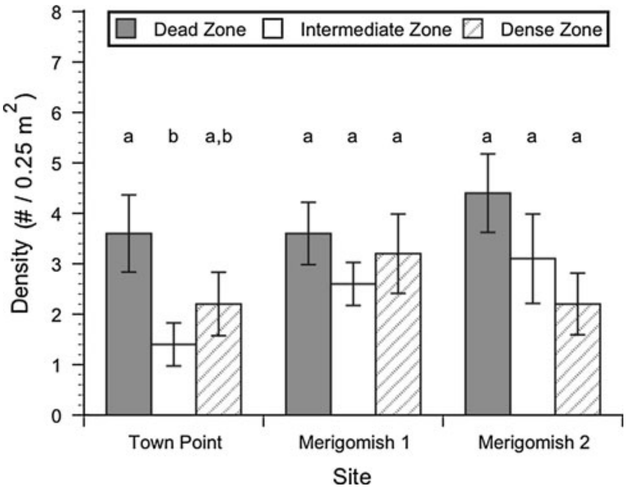

Fig. 5 Mean density (per 0.25 m^{-2}) of *Geukensia demissa* in each of the *Spartina alterniflora* zones at three sites. *Different letters* indicate significant differences. *Error bars* represent standard error; $n = 10$

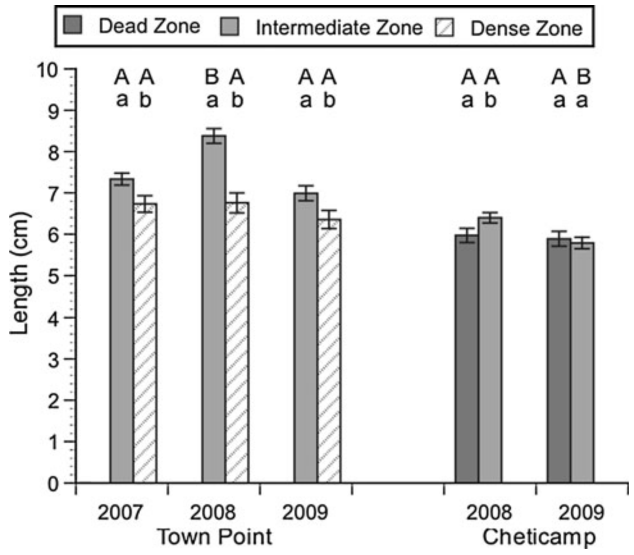

Fig. 6 Mean mussel length across the three *Spartina alterniflora* zones at Town Point in Antigonish Harbour in 2007 and 2008. *Different lowercase letters* indicate significant differences between zones within a single year ($P < 0.05$); *different capital letters* indicate significant differences within a zone between the 2 years sampled ($P < 0.05$). *Error bars* represent standard error; $n = 50$

Merigomish 2 and the sites in Pugwash Harbour (Tukey's test, $P < 0.05$) (Fig. 4).

Mussel density data were normally distributed and met the homogeneity of variance assumption (Levene's test, $F_{2,\,87} = 0.6044$, $P = 0.5487$). There was no significant difference among mussel density across the three sites (ANOVA, $F_{2,\,81} = 1.3608$, $P = 0.2622$), and there was no significant interaction between Site and Zone (ANOVA, $F_{2,\,81} = 0.7547$, $P = 0.5578$). However, there was a significant difference between mussel density across the three *S. alterniflora* zones (ANOVA, $F_{2,\,81} = 4.4428$, $P = 0.0148$), with the highest density occurring in the dead zone (Tukey's test, $P < 0.05$). However, when mussel density was examined at individual sites, the densities across zones were non-significant at both Merigomish sites and were only significant at Town Point (Fig. 5). At Town Point, density in the dead zone was significantly greater than the intermediate zone (Tukey's test, $P < 0.05$), but not significantly different from the dense zone (Tukey's test, $P > 0.05$) (Fig. 5).

The sediment samples taken at Town Point revealed no small mussels buried in the sediment. Upon return to Town Point in 2008 and 2009, the dead mats were reduced in size and had been drastically eroded compared to the previous year. At the sites in Pomquet Harbour in 2008 and 2009, the population of *Geukensia demissa* was extraordinarily small and after intensive searching for 30 min only 11 mussels were encountered; therefore, there were not sufficient mussels present to measure.

Mussel length was recorded at the Town Point site in Antigonish Harbour in all three *S. alterniflora* zones in 2007 and in only the intermediate and dense zone in 2008 and 2009; thus analyses over time were only conducted on the intermediate and dense zones. Mussel populations in the dead zone were not measured because the population was reduced to 15 and 13 mussels in 2008 and 2009, respectively, compared to >50 in 2007, therefore we considered sampling detrimental to their survival. Mussel length data over time were normally distributed but failed to meet the homogeneity of variance assumption (Levene's test, $F_{2,\,297} = 5.4953$, $P = 0.0045$); thus, data were arc-sin transformed to meet this assumption (Levene's test, $F_{2,\,297} = 3.1175$, $P = 0.05$). At Town Point, there was a significant difference in mussel length over time (ANOVA, $F_{2,\,294} = 6.6310, P = 0.0015$) and between zones (ANOVA, $F_{1,\,294} = 30.7523, P < 0.0001$), but no significant interaction between the two factors (ANOVA, $F_{2,\,294} = 2.22931$, $P = 0.1028$). Mussel length was greater in the intermediate *S. alterniflora* zone compared to the dense zone in all three years (*t*-test, $P < 0.05$) (Fig. 6). There was no significant difference in mussel length among years in the dense zone (ANOVA, $F_{2,\,147} = 1.0240$, $P = 0.3617$); however, there was a significant difference among years in the intermediate zone (ANOVA, $F_{2,\,147} = 18.4096, P < 0.0001$) with mussel length being greater in 2008 compared to 2007 or 2009 (Tukey's test, $P < 0.05$) (Fig. 6).

At the Cheticamp site, mussel length data over time were normally distributed and met the homogeneity of variance assumption (Levene's test, $F_{1,\,198} = 0.3772$, $P = 0.5398$). There was a significant difference in mussel length between the intermediate and dense zones in 2008

($t = 1.9976$, $P = 0.0485$), but no significant difference in 2009 ($t = 0.4391$, $P = 0.6616$) (Fig. 6). There was no significant difference in mussel length between 2008 and 2009 in the dead *S. alterniflora* zone ($t = 0.3211$, $P = 0.7489$), but there was a significant difference in the intermediate zone ($t = 3.2277$, $P = 0.0017$) (Fig. 6).

Discussion

This is the first study to investigate the distribution of *G. demissa* in marshes in the southern Gulf of St. Lawrence and represent the most northern populations studied to date. Our results show a difference in mussel length across zones of *Spartina alterniflora* in seven of the nine sites. Although differences in growth rate across the zones could have contributed to our findings of different mussel lengths across the *S. alterniflora* zones, there were no small mussels (1–3 cm) found in any zone, suggesting a lack of recent recruitment and an unsustainable population in these sites. Potential explanations include the following: (1) we were unable to locate the smallest mussels, (2) recruits are removed by predation, or (3) recruitment could be failing because of declining *S. alterniflora* in the dead mats where these mussels should be prospering.

In the dead and intermediate, zones of *S. alterniflora* mussels were easy to locate and were clumped together. Mussels in the dense zone were the hardest to locate because they were usually smaller, surrounded by dense *S. alterniflora*, usually solitary, and often buried deeply in the mud with 1 cm or less protruding. Even though mussels were hardest to locate in the dense zone, this is where we encountered the smallest mussels; thus, we believe if these mussels were located in the dead mats, we would have located them. Furthermore, the sediment samples revealed we were not overlooking very small mussels that may not be distinct in the field.

Increased predation could explain absence of recruits. A conspicuous predator is the invasive green crab, *Carcinus maenas*, which has had significant impacts on local populations of *Mya arenaria* (L.) (Floyd and Williams 2004) and may have contributed to the decline of *Zostera marina* in regional estuaries (Seymour et al. 2002). Although the population of *C. maenas* may be large enough to impact the abundance of young recruits on the marsh, we do not believe this is causing the absence of recruits. In 2007 and 2008 at all sites, in all harbors, populations of *Mytilus edulis* (L.) have been large with a wide range of mussel sizes present. If *C. maenas* was responsible for the complete absence of recruits, populations of *M. edulis* should also be affected because *C. maenas* is known to feed on *M. edulis* (Breen and Metaxas 2008). Preferential feeding on *G. demissa* is unlikely considering its low density in these marshes, and its significantly greater shell thickness compared to *M. edulis* (Alexander and Dietl 2001). Furthermore, *C. maenas* is also present in the two sites in Pugwash Harbour where at least a few small mussels were found. As a result, we suggest *C. maenas* cannot be solely responsible for the lack of recruits at the other sites.

A final possibility is that mussels are no longer recruiting to the marsh and have not been recruiting to the lowest marsh levels where they would normally be found in high abundances in healthy marshes for some time. Based on the size age distribution presented by Lutz and Castagna (1980), recruitment in seven of the nine sites, even to the dense marsh zone, has not been occurring for at least 2 years and has been extremely limited for at least three additional years. This may be a result of sea level rise. The integrity of salt marshes in Atlantic Canada and elsewhere is being jeopardized by increases in sea level (Najjarr et al. 2000; Hartig et al. 2002; Gehrels et al. 2005; Erwin et al. 2006; Garbary et al. 2008). Effective sea level in the southern Gulf of St. Lawrence has risen over 30 cm during the twentieth century and is predicted to increase at least another 70 cm this century (Forbes et al. 2004). Effective sea level includes changes from water level rise and land subsidence (Forbes et al. 2004). This will be exacerbated by even more rapid rates of polar ice melt than was predicted by earlier models (e.g., Velicogna and Wahr 2006). Sea level rise will affect many species, and marsh communities are particularly sensitive because many of the species within these communities cannot withstand high salinities for extended periods. Sea level rise may be the cause of the marsh edge erosion where *S. alterniflora* grows (Garbary et al. 2008). This may result in decreased recruitment of *G. demissa* because the marsh edge is where the mussels tend to settle (Bertness and Grosholz 1985). Recruitment of *G. demissa* at the marsh edge has been shown to significantly decrease in experimental plots where *S. alterniflora* has been removed (Bertness and Grosholz 1985). In our case, *S. alterniflora* has been naturally removed, and recruitment has not been occurring for a few years. Since *G. demissa* and *S. alterniflora* have a unique facilitative interaction in marsh systems (Bertness 1984), the reduction of one can result in decreased abundance of the other. The death of *S. alterniflora* at the marsh edge may have resulted in decreases in *G. demissa* recruitment to that area. In addition, the absence of new recruits to the marsh edge, and limited recruitment to the intermediate zone, may result in decreased production of *S. alterniflora* (Bertness 1984), leading to further decreases in the cordgrass throughout the entire marsh.

A few small mussels (1.0–3.0 cm) were found, albeit in small abundances, in the two sites located in Pugwash

Harbour, and mussel size was not significantly greater in the dead zone in these sites. We suggest these populations have experienced recruitment more recently than the other populations. The dead mats at these sites were still contiguous with the living marsh and were smaller in width compared to dead zones at the other sites. The delayed impact in these estuaries may be a result of harbor topography. Accordingly, the channel into Pugwash Harbour is much deeper than the other harbors, and it lacks a sand dune barrier beach, present in the other estuaries. These differences may lead to increases in sedimentation that could compensate for the eroding dead zone. Models presented by Gorand and Monaco (1994) suggest that varying current speeds can have a drastic impact on sedimentation rates. According to Jones (1994), the shape and size of the estuary, as well as the geology of the drainage basin feeding the estuary, are important factors that will affect how sea level rise will impact each estuary. We suggest future studies might evaluate sedimentation and erosion rates in these estuaries to determine how sea level rise is impacting each estuary to provide support for this hypothesis.

Regionally, sea level rise is predicted to increase at a faster rate than we have seen to date (Forbes et al. 2004). Previous studies have found that marsh area on the Atlantic coast of North America is decreasing (Hartig et al. 2002), and marsh surface elevation is not keeping pace with current sea level rise (Erwin et al. 2006). One model suggests that sea level rise will reduce intertidal area, leading to reductions in macro-benthic biomass of up to 12% with a sea level rise of 0.5 m (Fujii and Raffaelli 2008). If the *G. demissa* size distribution we obtained in this study is a result of increases in sea level rise, whether or not *S. alterniflora* and *G. demissa* can move further from the waters edge fast enough to establish populations prior to water levels becoming prohibitive, remains to be established. Due to the importance of both *S. alterniflora* and *G. demissa* in salt marshes in the northern Atlantic, we suggest future studies monitor populations closely and further investigate the association between sea level rise and diminishing cord grass. Jost and Helmuth (2007) suggested that *G. demissa* and its relationship with *S. alterniflora* would be a suitable system for monitoring impacts of climate change; this seems particularly the case in the southern Gulf of St. Lawrence.

Acknowledgments We are grateful to Eric Wheeler for field assistance and Ricardo Scrosati for comments on a draft of the manuscript. Cortney Watt was funded through the Canada Research Chair (CRC) and the Natural Sciences and Engineering Research Council of Canada (NSERC) Discovery Grant to R. Scrosati. Research was supported by NSERC grants to DJG.

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The Community Aquatic Monitoring Program (CAMP) for Measuring Marine Environmental Health in Coastal Waters of the southern Gulf of St. Lawrence: 2007 Overview

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2009

Canadian Technical Report of
Fisheries and Aquatic Sciences 2825

Fisheries and Oceans Canada Pêches et Océans Canada

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Canadian Technical Report of Fisheries and Aquatic Sciences

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February 2009

The Community Aquatic Monitoring Program (CAMP) for Measuring Marine Environmental Health in Coastal Waters of the southern Gulf of St. Lawrence: 2007 Overview

by

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Cat. No. Fs. 97-6/2825 ISSN 0706-6457

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Correct citation for this publication:

Weldon, J., Courtenay, S. and Garbary, D. 2008. The Community Aquatic Monitoring Program (CAMP) for measuring Marine Environmental Health in Coastal Waters of the southern Gulf of St. Lawrence: 2007 Overview. Can. Tech. Rep. Fish. Aquat. Sci. 2825: viii + 75 p.

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ABSTRACT

In 2003, the Department of Fisheries and Oceans (DFO) Gulf Region initiated the development of a monitoring program called the Community Aquatic Monitoring Program (CAMP). One of the program goals was to help determine the ecological health of estuaries and coastal shorelines in the southern Gulf of St. Lawrence (sGSL). The primary goal of CAMP continues to provide an outreach program for DFO to interact with community environmental groups. The monitoring portion of CAMP is being used to test the hypothesis that a relationship exists between the health of an estuary or coastal shoreline and the diversity and abundance of finfish and crustacean species which inhabit the intertidal and near shore zone. CAMP expanded the number of locations from 4 in its 2003 pilot year (Thériault et al. 2006) to 24 throughout the Maritime Provinces of Nova Scotia (NS), New Brunswick (NB) and Prince Edward Island (PEI) in 2004. Baseline sites, meaning sites at which 6 stations were sampled by day-time beach seining once a month from May to September inclusive numbered 13 in 2004. In 2005, the number of locations totalled 22 of which 20 were considered as baseline (Weldon et al. 2007). In 2006, there were 22 locations participating and 18 were able to collect data for the five full months. In 2007, the number of baseline sites increased to 25, 24 of which maintained baseline status. NGOs in each watershed adhered to the same sampling methodology and related protocols as outlined in Weldon et al. (2005). All species of finfish, crab and shrimp collected were identified, separated into adults and young of the year, enumerated and released. Habitat was also characterized by collecting information such as water temperature, salinity, dissolved oxygen, % plant cover and algae cover and, once a year in September, collection of a substrate sample for measurement of grain size distribution, % moisture content and % organic content. Two water samples were collected at each station at all locations all 5 months then sent away for analysis of nutrient content. This report summarizes baseline physical and biological data for the estuaries sampled in 2007. This year more almost six hundred thousand (597295) animals were processed and 37 different species were identified. In order to test the hypothesis that these data reflect environmental quality, we are getting close to the several years of data required to detect temporal and spatial patterns that may exist.

RÉSUMÉ

En 2003, le Ministère des Pêches et Océans (MPO) de la Région du Golfe a mis au point un programme de surveillance intitulé Programme de surveillance de la communauté aquatique (PSCA) afin d'évaluer la santé écologique des estuaires et des zones du littoral du sud du golfe du Saint-Laurent (sGSL). L'objectif primaire du PSCA continu toujours d'offrir un programme d'extension permettant au MPO d'interagir avec les groupes environnementaux des collectivités. L'aspect de surveillance issu de ce partenariat vise à mettre à l'essai l'hypothèse qu'une relation existe entre la santé d'un estuaire ou d'une zone côtière et la diversité et l'abondance de poissons et de crustacés qui se trouvent dans la zone côtière. Le PSCA est passé de 4 emplacements lors de l'année du projet pilote (Thériault et al. 2006) à 18 répartis partout dans les provinces Maritimes, soit la Nouvelle-Écosse (N.-É.), le Nouveau-Brunswick (N.-B.) et l'Île-du-Prince-Édouard (Î.-P.-É.) en 2004. Les emplacements principaux, soit les emplacements où on a effectué des prélèvements mensuels diurnes dans six stations, à l'aide de seines de plage, de mai à septembre, s'élevaient à 13 en 2004. En 2005, le nombre d'emplacements atteignait 22, dont 20 emplacements principaux. En 2006, 22 sites ont été échantillonnés dont 18 d'entres eux ont été échantillonnés de mai à septembre. Chaque groupe environnemental communautaire a utilisé la même méthode d'échantillonnage et les protocoles connexes décrits par Weldon et al. (2005). Les individus de chaque espèce de poissons, de crabes et de crevettes capturés à l'aide d'une seine de plage ont été énumérés, identifiés, triés selon l'âge (jeunes de l'année et adultes) puis remis à l'eau. De plus, des données sur l'habitat de ces espèces ont été recueillies telles que la température de l'eau, la salinité, la teneur en oxygène dissous et le pourcentage de recouvrement par les plantes et les algues une fois par mois. De plus, la distribution de taille des grains, le % de la teneur en eau et de la teneur en matières organiques du substrat ont été recueillies une fois par an, soit en septembre. Deux échantillons d'eaux ont également été collectés à chaque station et analysés pour déterminer le contenu en nutriment (nitrate, nitrite, ammoniac, phosphate et silicate). Le présent rapport résume les données physiques et biologiques des emplacements principaux des estuaires étudiés en 2006. Cette année, un peu moins de quatre cents milles animaux ont été comptés et 37 différentes espèces ont été identifiées. Pour pouvoir vérifier l'hypothèse selon laquelle ces données reflètent la qualité de l'environnement estuarien, plusieurs années de données devront être étudiées afin de détecter les tendances temporelles et spatiales qui pourraient exister. On espère que le programme s'avèrera une méthode simple de caractérisation de la santé estuarienne qui sera à la fois utile et facile à appliquer pour les groupes communautaires.

1.0 INTRODUCTION

1.1 Background

In the Canada Oceans Strategy document (COS, 2002), Fisheries and Oceans Canada (DFO) established its commitment to work collaboratively with local stakeholders to "*establish marine environmental quality guidelines, objectives and criteria respecting estuaries, coastal waters and marine waters.*" During 2003 and 2004, the Stewardship and Aquatic Ecosystem Sections of DFO Gulf Region integrated their planning priorities to develop a practical monitoring program that would assist in determining the ecological health of estuaries in the southern Gulf of St. Lawrence (sGSL) as outlined in Canada's Stewardship Agenda (2003). The outcome was the development of the Community Aquatic Monitoring Program (CAMP) outlined in detail in the first report (Weldon et al. 2005). One of the aims of the program is to determine if a relationship exists between the health of an estuary and/or a coastal shoreline and the diversity and abundance of conspicuous crustaceans and fish species which utilize this ecosystem. This concern over animal abundance is highlighted by a regional concern over the decline in eelgrass populations (Hansen 2004) and the dependency of many of the estuarine animals on eelgrass as a primary habitat. One hypothesis being tested by CAMP is that an estuary which has been degraded by human activity may have fewer species and different abundance of individuals than a healthy, undisturbed estuary.

Methods and protocols to implement the CAMP approach were chosen after reviewing a wide variety of methods for evaluating estuarine health and population dynamics (Karr 1981, Methven et al. 2001, Whitfield and Elliot 2002). Standardized methodology continues to be followed in 2007 (see Weldon et al. 2005). This report will provide an overview of the CAMP results in 2007 and very briefly discuss some of the similarities and differences with outcomes of the 2004 - 2006 field seasons. A subsequent report will provide a more in-depth comparison of five years (2004-2008) of CAMP data.

1.2 Where has CAMP taken place?

CAMP is a long term monitoring program used to determine the ecological health of estuaries and coastal shorelines in the sGSL region. To become a baseline location, an estuary or coastal shoreline would be sampled monthly during the spring and summer months (May - September) (5 times) at 6 chosen stations. Total baseline locations for the 2004 was 13, followed by an increase to 20 in 2005, then a decrease to 18 in 2006 (due to decreased capacity of some groups to complete full baseline) and additional expansion to 24 in 2007. Changes in 2007 included Cocagne becoming full baseline again and Shediac River being added in New Brunswick. Pugwash and River Philip each became full baseline locations in Nova Scotia. Summerside was added as a new baseline in Prince Edward Island and the 3 locations in southeast PEI (Montague - Brudenell R.; Murray R.; Pinette R.) returned to full baseline status.

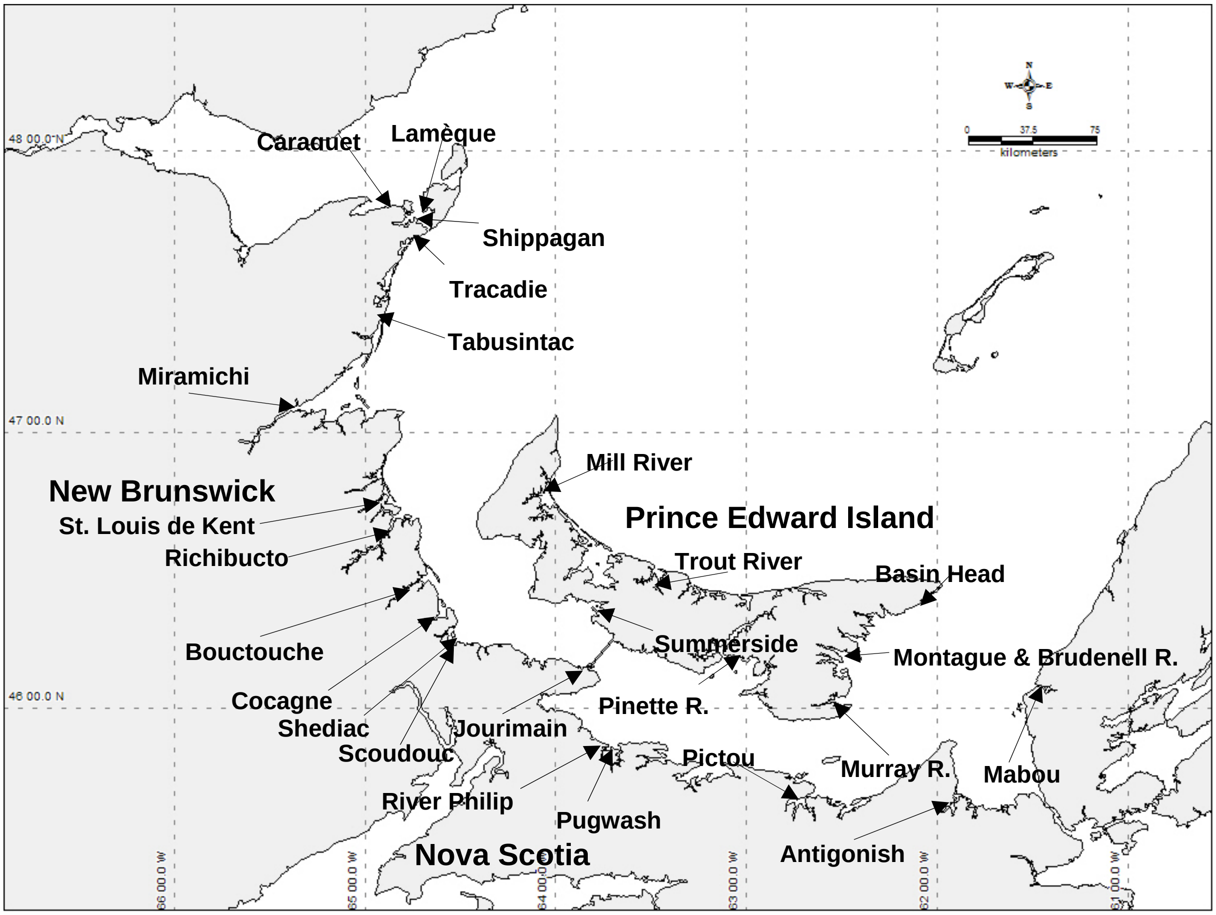


Figure 1. 2007 CAMP baseline monitoring locations for NB, NS and PEI. Each arrow shows the location of a baseline sampling estuary that includes six sample stations.

The CAMP program continues to involve several partnerships including DFO Oceans and Habitat and Environmental Science Divisions, DFO Area offices, Universities, various environmental organizations and local estuary community watershed groups, all based throughout the Gulf Region.

The groups who participated in 2007 include:

<u>New Brunswick</u>

- Partenariat pour la gestion intégrée du bassin versant de la baie de **Caraquet**
- Coalition pour la viabilité de l'environnement des havres de Shippagan et les Îles Miscou et Lamèque (**Lamèque and Shippagan**)
- L'association des bassins versants de la Grande et Petite rivière **Tracadie**
- **Tabusintac** Watershed Association
- **Miramichi** River Environmental Assessment Committee
- Friends of the Kouchibouguacis River (**St. Louis de Kent**)
- Elsipogtog First Nation Fisheries Management (**Richibucto**)

- Southeastern Anglers Association (**Bouctouche**)
- Pays de **Cocagne** Sustainable Development Group
- Shediac Bay Watershed Association (**Shediac and Scoudouc**)
- **Cape Jourimain** Nature Centre

Prince Edward Island

- **Mill River** Watershed Improvement Committee
- **Trout River** Environmental Committee
- **Basin Head** Lagoon Ecosystem Conservation Committee
- Southeast Environmental Association (**Pinette, Montague-Brudenell, Murray**)
- Montague Watershed Improvement Committee
- Bedeque Bay Environmental Management Committee (**Summerside**)
- Students from the University of Prince Edward Island Biology Department

Nova Scotia

- Friends of the **Pugwash** Estuary
- Cumberland County Rivers Association (**River Philip**)
- Fresh Air Outdoor Adventure Society (**Antigonish**)
- **Mabou** Harbour Coastal Management Planning Committee
- Students from the St. Francis Xavier University Biology Department and program in Integrated Studies in Aquatic Resources (ISAR) (**Pictou**)

Participation of community groups is a fundamental strength of the CAMP program, as NGO's share the responsibility of volunteering their time to monitor estuaries and coastal shorelines in their area. As NGO's often have several projects related to the estuary, their work is fundamental in demonstrating and initiating efficient stewardship principles.

2.0. MATERIALS AND METHODS

Monthly daytime sampling was done from May to September for the baseline sites. In this report a location (or site) refers to the specific estuary or coastal sample area and a station refers to one of the 6 areas at each site where beach seines were used to sample the shoreline community. Data on crustaceans and fish species, macrophytes, water quality and benthic substrate were collected at 150 baseline sampling stations throughout the provinces of NB, NS and PEI. Physical data included the use of a quadrat for vegetation cover, YSI meter readings to record temperature, salinity and dissolved oxygen, a sediment collection and a water sample collected for nutrient analysis. Methodology and protocols are described in more detail in Weldon et al. (2005).

2.1 Training

The new training and refresher review program for CAMP participants takes place in May and is a combination of theory and practical sessions. The theory session consist of an hour long presentation on CAMP which includes background, an outline of the methodology, an introduction to the equipment, training on use of the field data collection sheets and a review of the identification of species sampled during the field season. Each year, the training regime is modified depending on the experience of past NGO coordinators and the need to train new employees and/or volunteers.

The practical session involves training to familiarize participants with the proper use of equipment, standardized techniques and proper identification of fish, crustaceans and plant species with an actual on-station beach seine collection. After a beach seining, the volunteers identify the contents of the beach seine while DFO trainers and NGO coordinators assist with verification and identification. A folder for plant and animal identification with the most commonly encountered species, mostly referenced from Scott and Scott (1988), was prepared and provided to each community group

2.2 Site Selection

In 2007, NGO's returned to the same estuary or coastal location to repeat the sampling regime at the same stations of the past year. There was some site movement for safety reasons but they usually only a few meters to the left or right of the original site. New baseline estuary sites were added in 2007 (Shediac River and Summerside) and 4 other locations regained full baseline status using protocols identical to those of the 2004 - 2006 collections (Weldon et al. 2004).

2.3 Fish Identification

When difficulties with identification arose, groups could refer to the CAMP identification guide for clarification, or collect a specimen for identification. One option was to get the unknown plant or animal back to the local Area Office or DFO HQ for identification. A species that could not be identified in the field would be put on ice and frozen or otherwise preserved upon return to their NGO office. Guides such as Peterson Atlantic Coast Fishes and Atlantic Seashore field guides were made available and distributed to community groups to be used to assist with identification. Groups were also encouraged to take a quality digital photograph to assist with later identification for any unknown species.

2.4 Substrate Characteristics

Each visit to a site at each location involved recording a percentage of what the volunteer considered the bottom to be composed of. The four main choices are sand, mud, gravel and rock; rock descriptions could vary in that they could be solid, have gravel or small stones or some combination of any of the above. After five visits to the site each season and because of varying tides, a volunteer could see entirely different bottom characteristics each visit. This is the reason that volunteers are encouraged to visit a site at similar tidal times each month. To get the best idea of the site bottom structure, the results will present an overall average of what was recorded for the five months. An even better picture of the site could emerge if all five years of observations were summarized

2.5 Substrate Composition

In September, a sample of benthic substrate was collected from each sample station to analyze moisture content, organic content and grain size distribution of the sediment. Using a garden trowel, a sample of the surficial ten cm layer was obtained from within the seine area, bagged and returned for freezing at -20˚C and stored for later analysis at the Gulf Fisheries Center. From each frozen sediment sample, a thawed portion (100 ± 20 grams) was removed and placed in an aluminum pan.

In the laboratory, moisture content was determined as the difference in weight before and after drying at 70°C for 24 h (standardized time). Organic content was calculated as the difference in weight before and after burning the sediments in a muffle furnace at 500°C for one hour (standardized time). Grain size distribution per sample was determined from 10 min shaking (standardized time) with a mechanical sieve shaker with six different sieve sizes: >2 mm (very coarse sand), >1 mm (coarse sand), >500 µm (medium sand), >250 µm (fine sand), >125 µm (very fine sand), >63 µm (coarse silt) and <63 µm (silt) (Higgins and Thiel 1988). The mean grain size (MGS) was then calculated from the cumulative frequency curves established with the grain size distribution.

2.6 Macrophyte Cover

A 50 cm x 50 cm quadrat, divided into four equal sub-quadrants, was used to estimate macrophyte percentage cover at each sample station. The quadrat was thrown three times, across the sample area from left to right at middle depth. The data sheet was used to record the approximate percentages of the dominant plant and algal types. The use of this quadrat method was possible only when the water column was not turbid. Wind and wave action stirred up the sediment and made the percentage cover evaluation difficult in some cases.

Volunteers also included a general description of the sample area by taking notes of the overall dominant macrophytes present, their approximate cover percentage and location in reference to the shore.

2.7 Physical Measures

YSI meter model 85 was used to measure three physical components of water: temperature (± 0.1 C°), dissolved oxygen (± 0.1 mg/L) and salinity (± 0.1 ppt). Meter readings were taken either before (adjacent to the sweep site) or after the beach sweep (within the net area). The YSI probe was submerged approximately at mid-depth in the vicinity of the center of the sample area.

Also in 2007, Vemco continuous temperature monitoring probes were deployed for most of the sampling season. This was done in 2005 by DFO personnel with a NGO representative or with Area Office coordinator assistance. In 2006, NGOs were given the probes to place within their estuary or coastal location. In 2007, NGO’s or DFO personnel deployed the probes attached to a wooden stick that was attached to a fixed structure such as a wharf or dock so the probe was one meter below the low water mark. Recovery was successful for about 60% of the probes.

2.8 Nutrient Analysis

In 2007, each group was supplied with 60 water bottles (30 ml) to collect two replicate water samples at each station for each month in their location. All 60 bottles were numbered and on the date of collection, matched with the station. The samples were collected on the sampling day and stored in a cooler bag with ice. Upon return to home base, they were frozen to await transport back to DFO where they were then sent to Halifax for analysis. The Bedford Institute of Oceanography physical parameter section (nutrient lab) completed the chemical analysis for nutrient content (ammonia, nitrate, nitrite, phosphate and silicate) during the winter. Results were made available in the spring of 2008. Detailed protocols for the treatment of the water samples is presented in Theriault and Courtenay, 2008 (unpublished report).

2.9 Permits

Each group was able to apply on-line to acquire a species sampling permit for scientific purposes. These are available from DFO Gulf region at the following location:

https://www.glf.dfo-mpo.gc.ca/fam-gpa/bssp-saps/s52/form-e.php?form_lgE=e

Persons listed as part of the community group on the Section 52 permit are authorized to collect, count and release fish species commonly found in estuarine locations. The permit also allowed them to collect and transfer unidentified specimens that required further identification.

3.0. RESULTS AND DISCUSSION

Descriptive statistics were used to examine the CAMP data and determine the relative abundance and species richness for sampling stations at all locations in each of the Maritime Provinces.

The total abundance values were determined by adding the totals for each species for the five months sampled of the season at each CAMP estuary or coastal site. Abundance of a particular species or grouped species of invertebrate or fish can be compared across sites and stations. Species richness was calculated by determining the total number of different species captured at each of all six stations located within a CAMP location, for each month sampled. In addition, the species richness was averaged across all stations for all five months sampled at each baseline site. Presenting the data in this way allows for comparisons among all the estuarine and coastal shoreline sample sites. Species richness graphs were therefore presented as a mean for the month. This information is also available in graphic form on posters developed for each geographical region of the Northumberland Strait.

3.1 Fish and Crustaceans

This section will discuss sampling results for locations with four to five complete months of sampling data. This includes the provinces of NB, NS, and PEI for the following 25 sites; Jourimain, Scoudouc, Shediac, Cocagne, Bouctouche, Saint Louis de Kent, Richibucto, Miramichi, Tabusintac, Tracadie, Lamèque, Shippagan, and Caraquet (NB); River Philip, Pugwash, Pictou, Antigonish, and Mabou (NS); and Mill River, Trout River, Basin Head, Summerside, Pinette River, Montague-Brudenell Rivers and Murray River (PEI).

For the 2007 sample season, a total of 597,295 adult and Young of Year (YOY) fish and crustaceans were counted from 13 baseline estuaries/coastal shorelines within NB, 5 in NS and 7 in PEI. Total adults numbered 478,722 and these numbers were used to produce graphed comparisons.

There were 34 different species of fish and crustaceans identified during the 2007 sample season, 28 of those species were fish and six were crustaceans. Species in **Appendix 1** are a list of those found in 2007 and include three invertebrate tunicates.

The five common species of crustaceans were; sand shrimp (*Crangon septemspinosa)*, grass shrimp (*Palaemonetes vulgaris*), rock crab (*Cancer irroratus*), green crab (*Carcinus maenas*), and mud crabs (*Xanthidae sp*.).

The most abundant fish and crustacean species were very similar for New Brunswick, Nova Scotia and Prince Edward Island. They were, in order of most abundant; sand shrimp (*Crangon septemspinosa*), both mummichog (*Fundulus heteroclitus*) and killifish (*Fundulus diaphanus*) grouped as *Fundulus* sp. (the majority of which were always mummichogs), 4-spine stickleback (*Apeltes quadracus*) and Atlantic silverside (*Menidia menidia*).

The ten most abundant species or species groups for each province/zone are graphed below (Figures 2 - 6B) to show the abundance relationship among the three sections in New Brunswick, namely northeastern (Figs. 2 & 2A), central (Figs. 3 & 3A) and southeastern sites (Figs. 4 & 4A), Nova Scotia (Figs. 5 & 5A) and Prince Edward Island sites (Figs. 6, 6A & 6B). Within the ten categories of species, the ‘other’ category pools the remaining less abundant species which sometimes represents a large number of certain species at specific stations at specific times of the season (eg. pipefish, smelts, striped bass). Because groups have collected up to 34 different species, the decision was made to group less numerous individuals rather than try to illustrate 34 graphs (Weldon et al. 2005). For each month, average species richness per beach seine haul was calculated across the six stations. The mean and 95 % confidence interval for these monthly estimates of species richness (SR) were plotted for each estuary or coastal location.

The legend for each graph (Figures 2 - 6) is similar in this report as it has for the previous two publications (Weldon et al. 2005, 2007). In the legend, “Stickle” refers to stickleback and “Killi” refers to killifish.

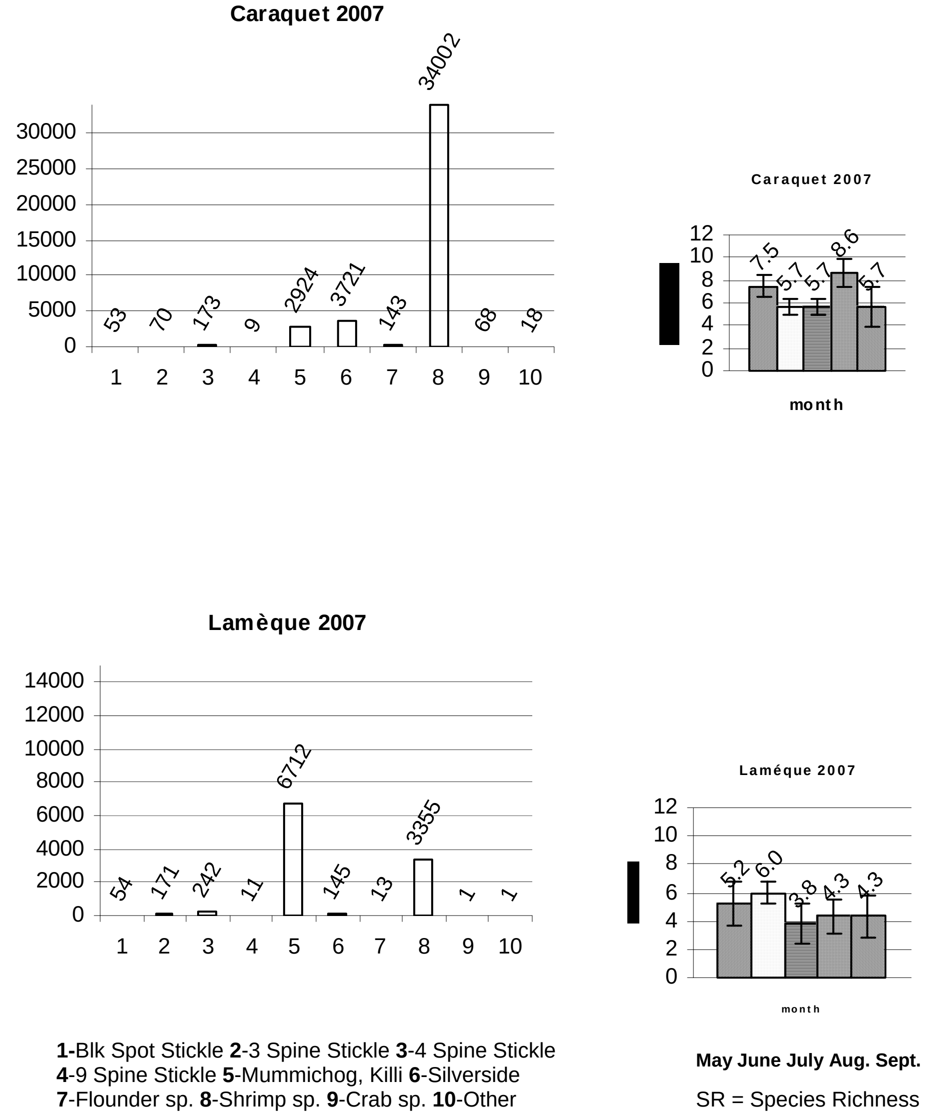


Figure 2. Season totals of adult animals for the 10 most abundant species or species groups in Caraquet and Lamèque (NB) sampled over 5 months in 2007. Mean taxon richness (total number of species taxa) is also shown for each month (95 % confidence interval CI) (n = 6 stations per bar).

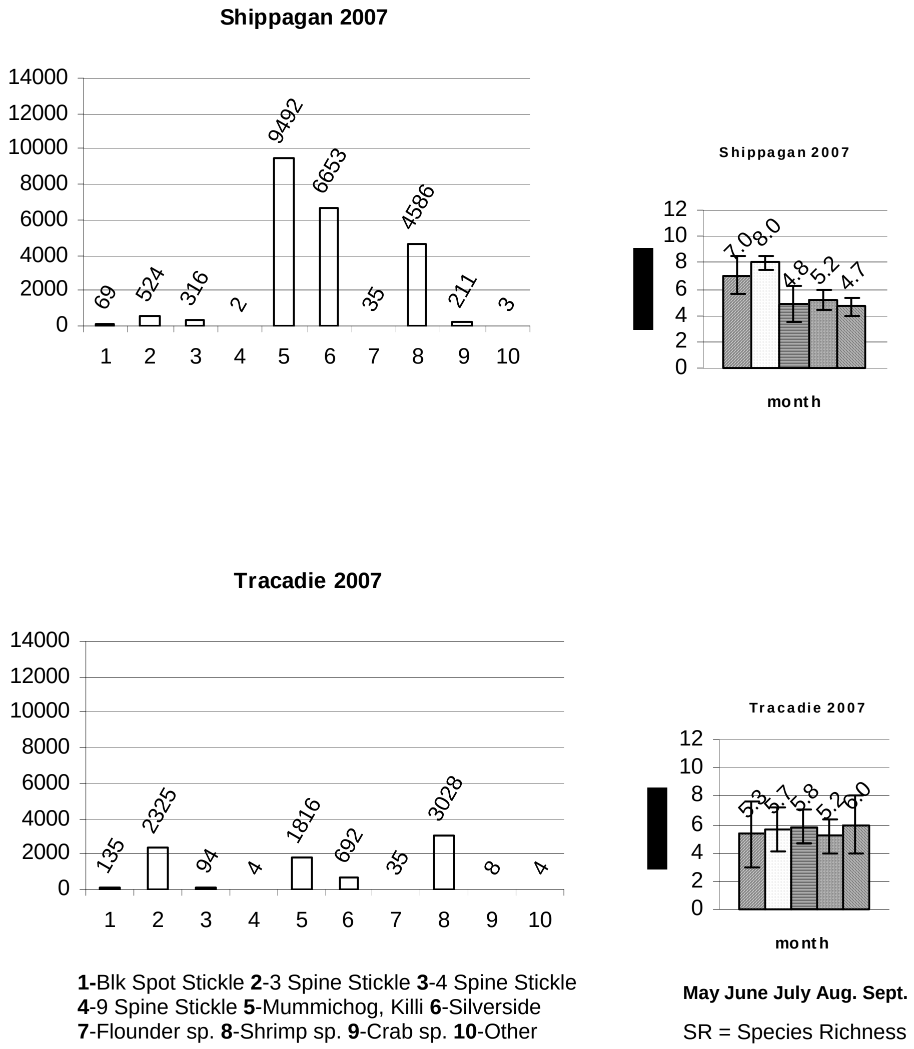


Figure 2A. Season totals of adult animals for the 10 most abundant species or species groups in Shippagan and Tracadie (NB) sampled over 5 months in 2007. Mean taxon richness (total number of species taxa) is also shown for each month (95 % confidence interval CI) (n = 6 stations per bar).

In northeastern NB (Figures 2 & 2A), the five most abundant species in Caraquet, Lamèque, Shippagan and Tracadie were mummichog, shrimp, silversides, three and fourspine sticklebacks. *Fundulus* species were the most abundant at Shippagan and Lamèque where shrimp was the most abundant at Caraquet and Tracadie. The threespine stickleback counts were higher than last year in two locations (Shippagan and Tracadie) while fourspine stickleback counts were higher in Lamèque and Caraquet. Ninespine sticklebacks remained at lowest total number of all the sticklebacks as it has for all years since 2005. Except for Shippagan in 2006, the black spotted stickleback numbers have generally decreased every year since 2005 in all locations in the northeast. The other two stickleback species, three and fourspine have shown season total numbers going up and down with no distinct trends obvious over the last four years.

In 2007, Tracadie had the lowest overall total number (8141) of adult fish and crustaceans for the season compared to the other three locations whereas Shippagan had the lowest totals (6601) in 2006. Caraquet and Shippagan had big total number increases compared to the previous year (41181 from 12628 and 21891 from 6601 respectively). These increases were mostly influenced by increased numbers of sand shrimp in Caraquet and by mummichog and silversides in Shippagan. With total numbers going up and down over the three - four years, many locations have no discernable pattern. Having multiple year data available may allow for specific patterns to be determined.

Generally, mean species richness was higher in June than it was in May, except in Caraquet where the trend was similar to 2006 with higher values in May than June. Tracadie and Caraquet had values for species richness slightly higher that last year while Shippagan and Lamèque maintained similar numbers to last year. It would appear that normal aggregative behaviour of these smaller pelagics as it relates to feeding and breeding may have been slightly delayed, possibly due to lower spring water temperature. As in previous years, the species richness declined slightly as the sampling season progressed with one notable exception in Caraquet in August where a high value of nine species was recorded. Analysis of five years of CAMP data is scheduled for completion in 2009 at which time trends in species richness may provide insights to certain population changes.

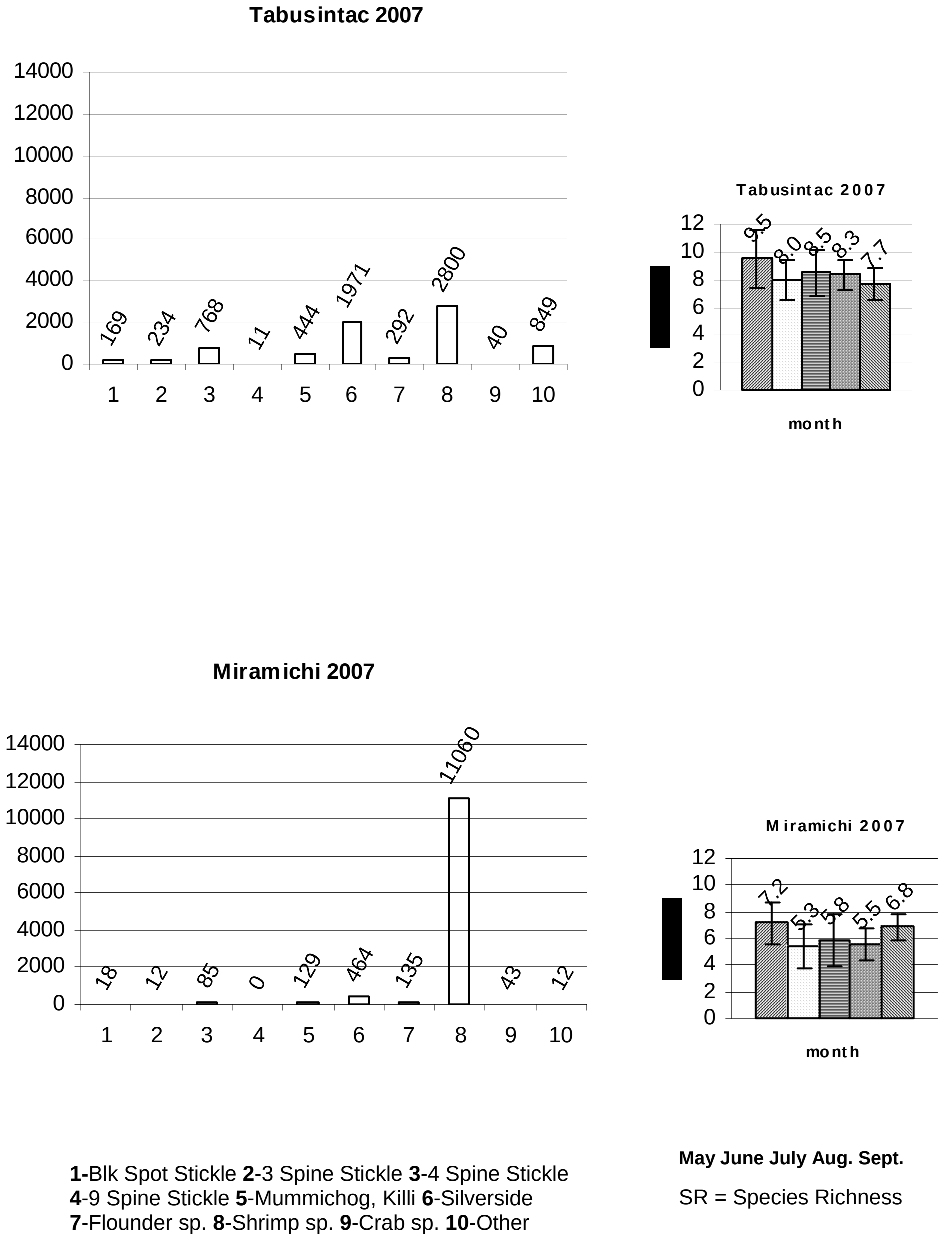


Figure 3. Season totals of adult animals for the 10 most abundant species or species groups in Tabusintac and Miramichi (NB) sampled over 5 months in 2007. Mean taxon richness (total number of species taxa) is also shown for each month (95 % confidence interval CI) (n = 6 stations per bar).

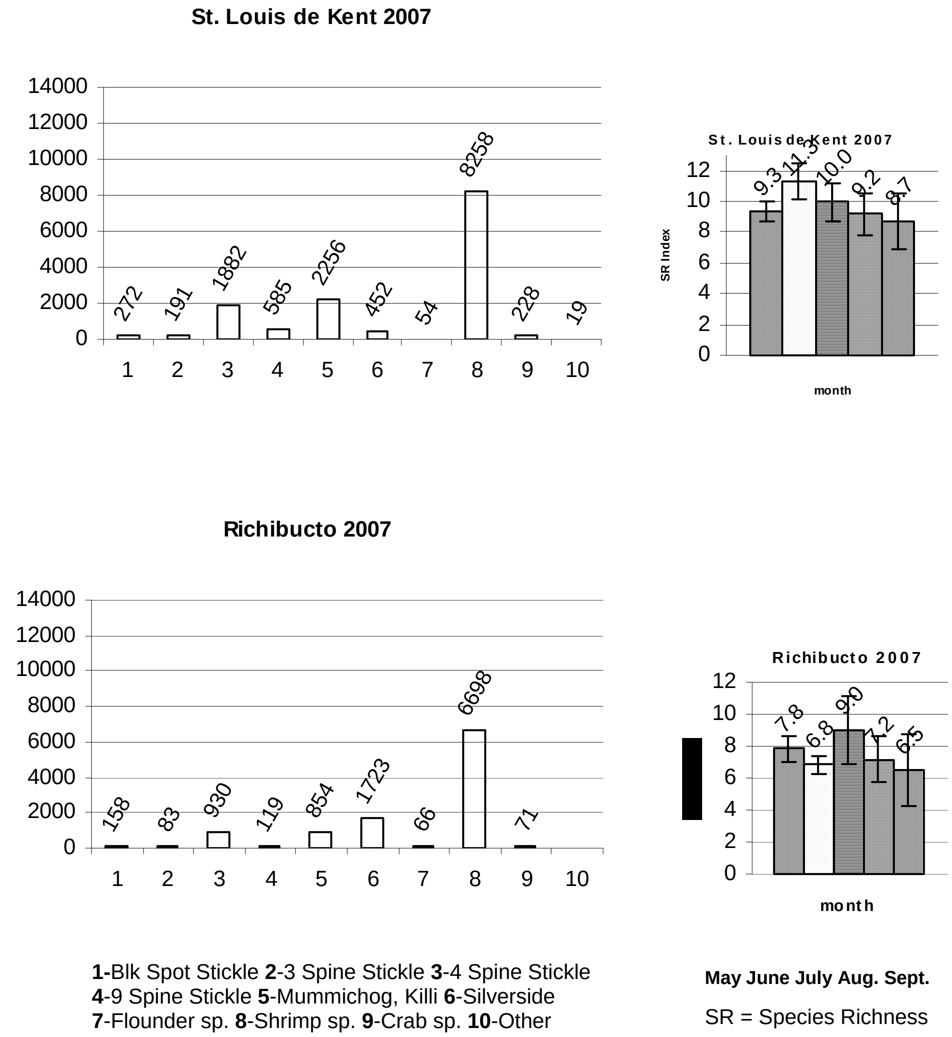


Figure 3A. Season totals of adult animals for the 10 most abundant species or species groups in St. Louis de Kent and Richibucto (NB) sampled over 5 months in 2007. Mean taxon richness (total number of species taxa) is also shown among the months (95 % confidence interval CI) (n = 6 stations per bar).

In central NB (Figure 3 & 3A), Tabusintac and Richibucto who joined Miramichi and St. Louis de Kent (Kouchibouguacis) as the baseline locations in 2006 maintained full status in 2007. Shrimp species were numerically the most abundant species in all four locations in 2007. Silversides were second in abundance totals for all fish species in all locations except St. Louis where mummichogs were more numerous than silversides. In the stickleback category, fourspine sticklebacks lead abundance numbers in all locations, a similar trend as shown in 2006.

In 2007, Tabusintac had the lowest overall total number (7578) for the season compared to the other three locations whereas Richibucto had the lowest totals (7076) in 2006. After a drop in 2006, St. Louis (14197) recorded an increase in 2007 for total species recorded as did Richibucto (10704). As in 2006, the higher abundance totals for species collected in St. Louis probably reflects the differences in estuary site characteristics. More vegetation, thus more habitats for protection, characterizes most of the St. Louis sites. Compared to other locations where sand is the more dominant substrate, more vegetation can help explain the higher abundance totals in St. Louis. Tabusintac and Miramichi had total number (11227 to 7578 and 138441 to 11958 respectively) decreases compared to the previous year. Patterns such as these may relate to certain factors in the environment and this will be explored in the five year summary report due in 2009. This variation could also be due to sampling at different times during the tidal regime.

In the “other” category, striped bass YOY do not show up as the abundance results are based on totals for adults only. In Miramichi alone, 2277 YOY striped bass were caught mostly in August. These were likely the result of a strong YOY spring survival as determined in the field (Scott Douglas pers. comm.). In previous years the YOY were probably incorrectly identified as juveniles who would have been put in the young adult category, though in most cases it has been determined they were young of the year. For other species, volunteers were instructed to put yearlings and juveniles who are not considered young of the year into the adult category.

Mean species richness showed no pattern with monthly numbers highest in May (Tabusintac), May and September (Miramichi), June (St. Louis) and July (Richibucto). A similar trend to 2005, 2006 data was evident in that species richness was higher overall in central NB when compared to northeast NB. Compared to last year, the overall species richness values was down in Miramichi and Richibucto, stayed about the same in St. Louis and was up slightly in Tabusintac in 2007.

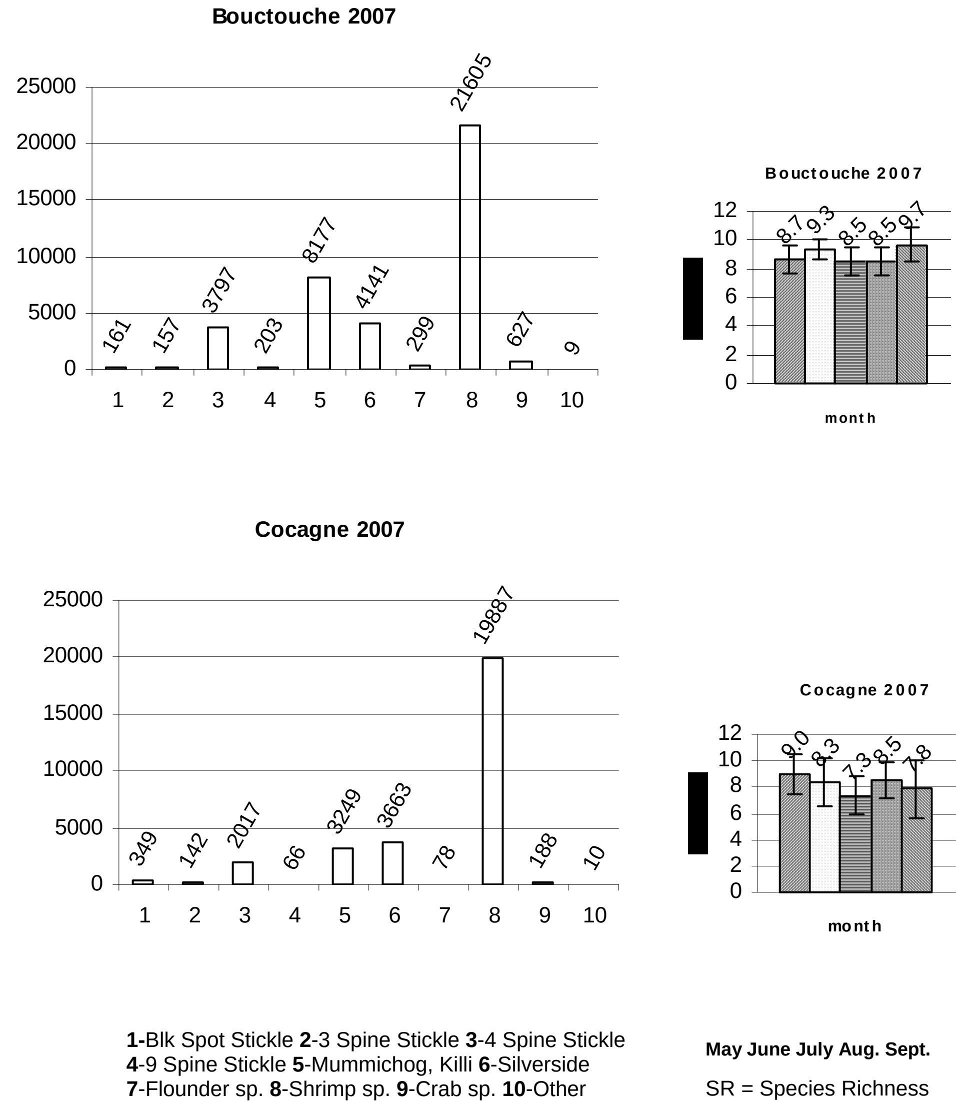


Figure 4. Season totals of adult animals for the 10 most abundant species or species groups in Bouctouche and Cocagne (NB) sampled over 5 months in 2007. Mean taxon richness (total number of species taxa) is also shown for each month (95 % confidence interval CI) (n = 6 stations per bar).

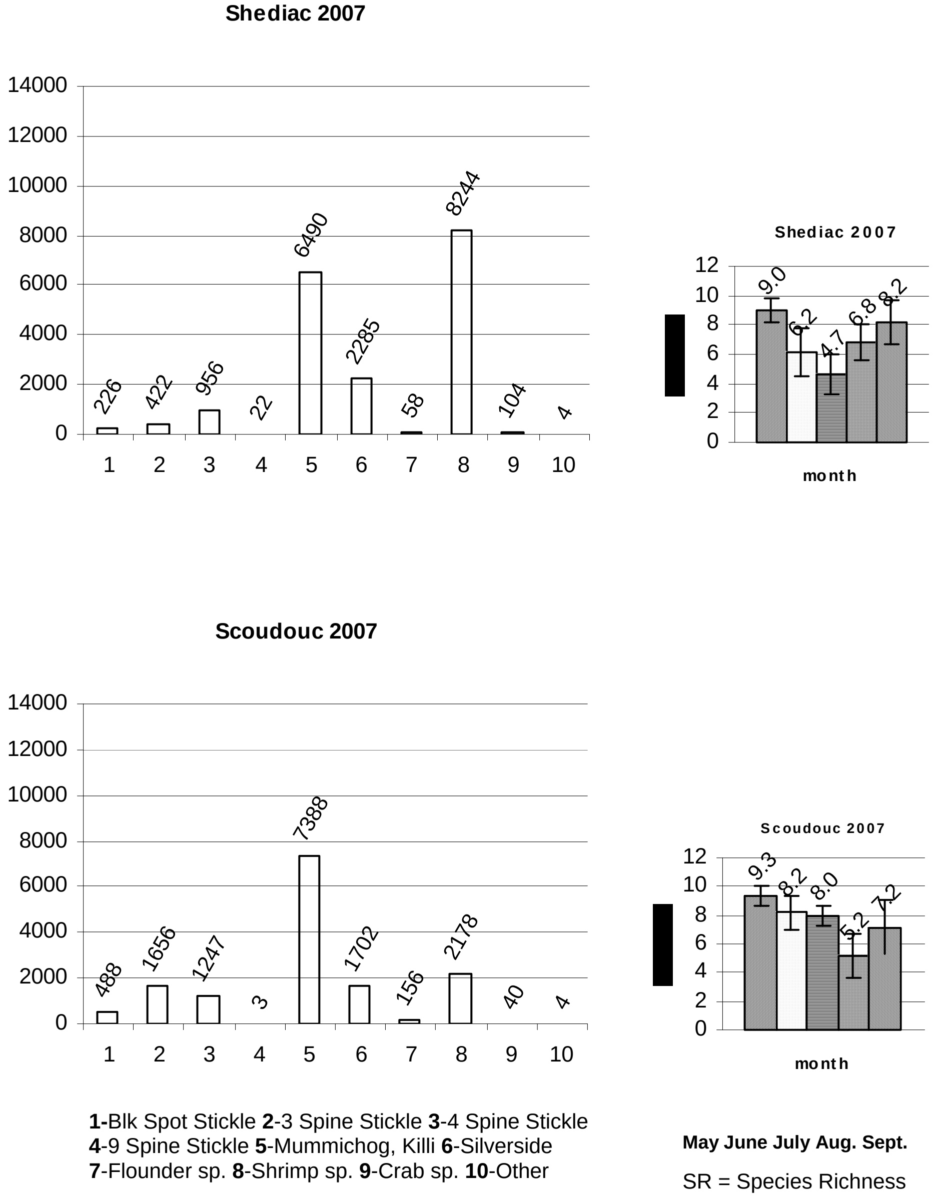


Figure 4A. Season totals of adult animals for the 10 most abundant species or species groups in Shediac and Scoudouc (NB) sampled over 5 months in 2007. Mean taxon richness (total number of species taxa) is also shown for each month (95 % confidence interval CI) (n = 6 stations per bar).

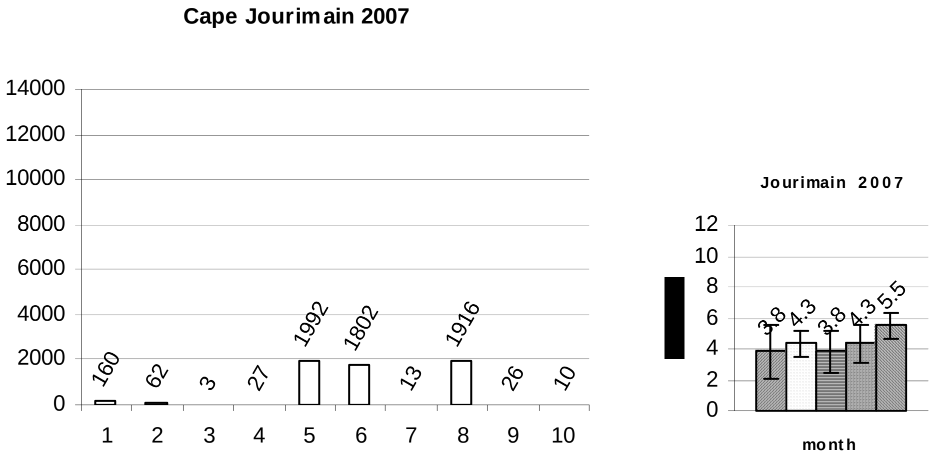


Figure 4B. Season totals of adult animals for the 10 most abundant species or species groups in Cape Jourimain (NB) sampled over 5 months in 2007. Mean taxon richness (total number of species taxa) is also shown for each month (95 % confidence interval CI) (n = 6 stations per bar).

In southeastern NB (Figure 4, 4A & part of 4B), Bouctouche, Scoudouc and Cape Jourimain maintained while Cocagne regained baseline status. The Shediac River became a new baseline site. Local NGOs know the difference but others should note for comparison purposes, and in the future, the Scoudouc River is in Shediac and the Shediac River is in Shediac Cape. Earlier reports refer to Shediac sites that are on the Scoudouc river system. Silverside numbers were highest in number for the fish in Cocagne and mummichogs were the abundant fish at the other four locations. Shrimp species were the most abundant crustacean and highest in species abundance everywhere except Scoudouc River. Interestingly, shrimp numbers have declined in the Scoudouc location over the last three years. Excluding Shediac River, the other four locations illustrate a shrimp number drop in 2006 and subsequent increase in 2007. Among the stickleback species, the fourspine was highest in abundance at three locations (Bouctouche, Cocagne and Shediac rivers). The ninespine sticklebacks showed increases everywhere except in the Scoudouc location. Cape Jourimain has maintained black spotted sticklebacks with the highest abundance totals as has been the case for the last three years. Cape Jourimain is more of a coastal sample area compared to the other locations (being estuaries) which might explain the lower overall abundance totals per species and higher blackspotted numbers. Though Scoudouc had the higher overall total (43156) for 2006, Bouctouche had the high total number (39176)of species in 2007. Of interest is total abundance numbers for Scoudouc dropped and in Bouctouche they increased by approximately 50% respectively from the previous year (2006). Cocagne total number of species was up from 21183 to 29649 and Jourimain dropped a bit from 6011 from 10139.

Mean species richness (SR) in the southeast was slightly less than central NB but greater than northern NB as it was the trend in previous years. The Bouctouche sites had the higher SR values compared to the lowest values for the coastal Cape Jourimain location. Species richness values fluctuate throughout the monthly sampling in all locations. Except for the Scoudouc River, all locations had lowest mean SR values in July. Compared to last year, Scoudouc stations were up, Bouctouche showed a slight drop, Jourimain a bigger drop and Cocagne remained about the same for monthly mean species richness values.

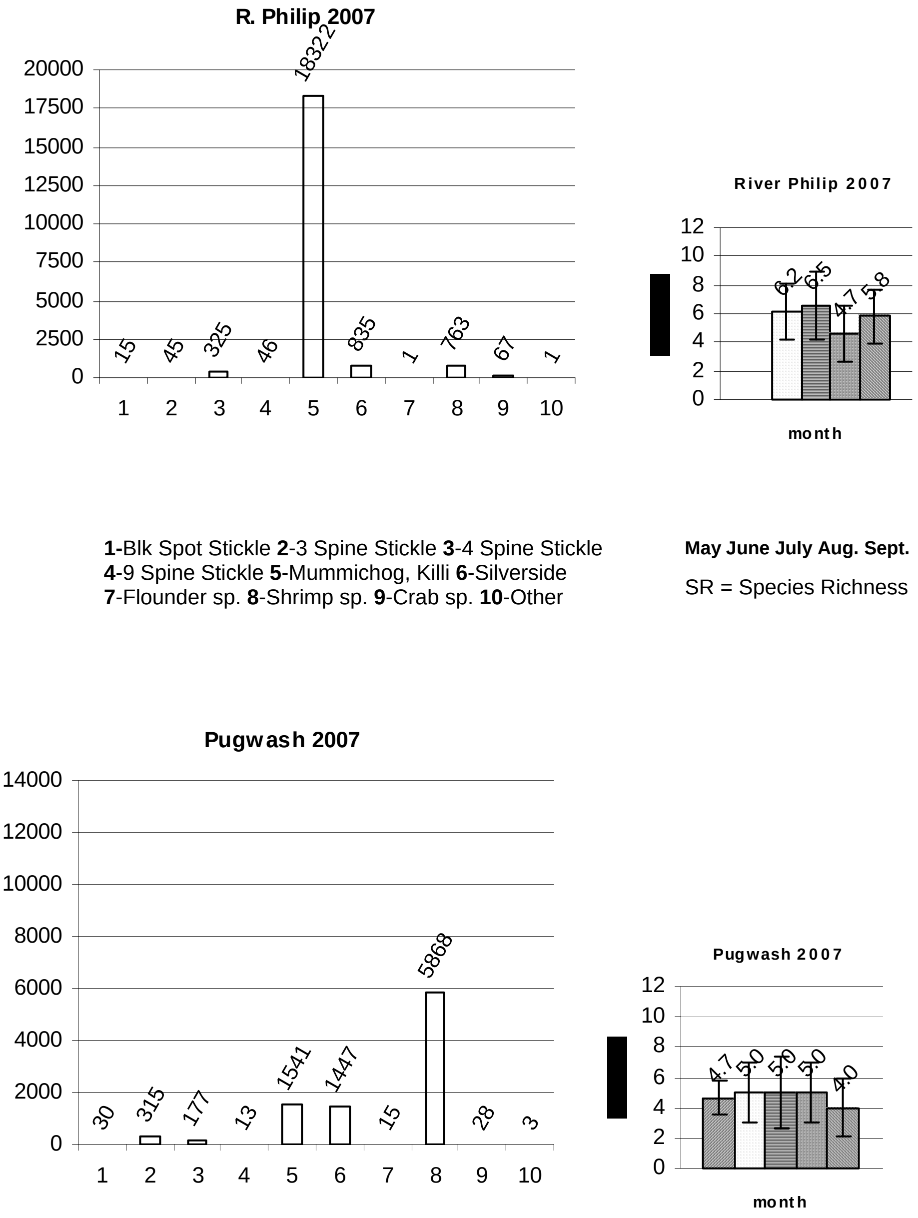
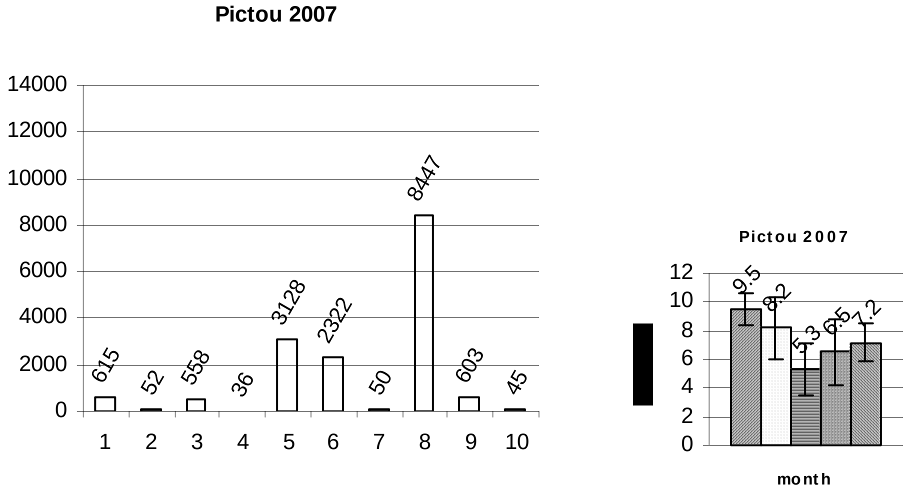


Figure 5. Season totals of adult animals for the 10 most abundant species or species groups in R. Philip and Pugwash (NS) sampled over 5 months in 2007. Mean taxon richness (total number of species taxa) is also shown for each month (95 % confidence interval CI) (n = 6 stations per bar).



1-Blk Spot Stickle **2**-3 Spine Stickle **3**-4 Spine Stickle
4-9 Spine Stickle **5**-Mummichog, Killi **6**-Silverside
7-Flounder sp. **8**-Shrimp sp. **9**-Crab sp. **10**-Other

May June July Aug. Sept.

SR = Species Richness

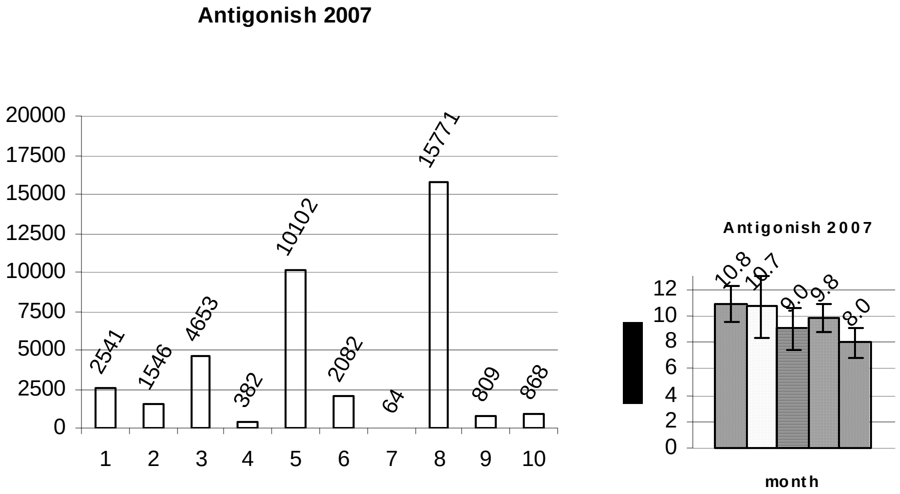


Figure 5A Season totals of adult animals for the 10 most abundant species or species groups in Pictou and Antigonish (NS) sampled over 5 months in 2007. Mean taxon richness (total number of species taxa) is also shown for each month (95 % confidence interval CI) (n = 6 stations per bar).

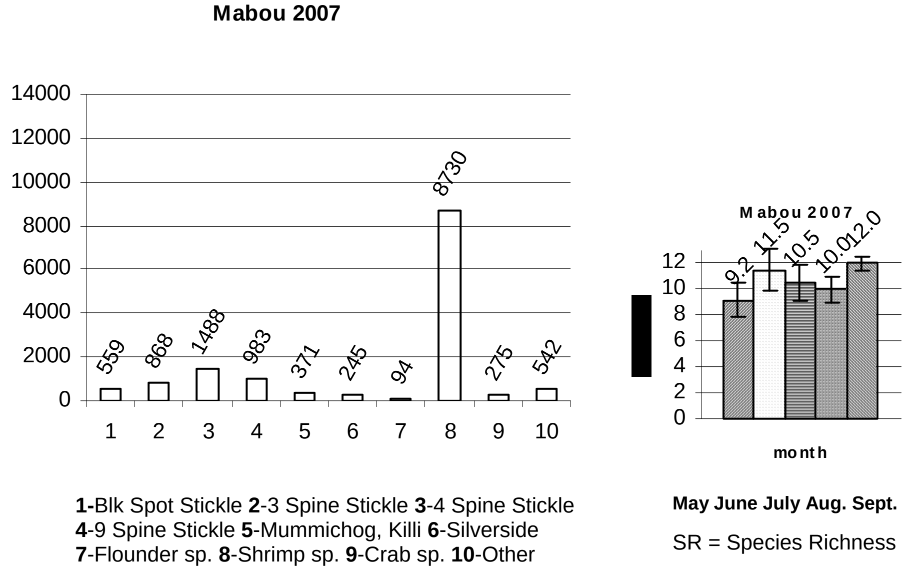


Figure 5B. Season totals of adult animals for the 10 most abundant species or species groups in Mabou (NS) sampled over 5 months in 2007. Mean taxon richness (total number of species taxa) is also shown for each month (95 % confidence interval CI) (n = 6 stations per bar).

In Nova Scotia, Pictou, Antigonish and Mabou maintained baseline status in 2006 (Figure 5 & 5A). These locations maintained a full sampling schedule because of regular assistance of biology students from St. Francis Xavier University. The River Philip-Pugwash combined location (in 2006) expanded in 2007 so each became a full baseline location, each with six stations or sampling sites.

Sand shrimp were highest in numbers in four locations, (Pugwash, Pictou, Antigonish and Mabou) but much lower in the newly established R. Philip site where *Fundulus* were most abundant. *Fundulus* species were second in abundance in Pugwash, Antigonish and Pictou. Silversides were second in abundance at R. Philip and Pugwash and fourspine sticklebacks second in Mabou. The most abundant stickleback species was the fourspine in three locations, Mabou, Antigonish and R. Philip. Black spotted sticklebacks had the highest total number in Pictou while threespines dominated in Pugwash. Among the four stickleback species, the abundance patterns when compared to the previous year changed at every location.

There were some large fluctuations in species richness in Nova Scotia throughout the sampling months. Overall comparisons of mean species richness illustrate lower values (5 - 7.5) for R. Philip-Pugwash to a higher range of 10 to 13 for Mabou. The others were in-between and the average for Nova Scotia was approximately seven. Compared to 2006, overall species richness for all five locations shows very small increases or very small decreases in 2007.

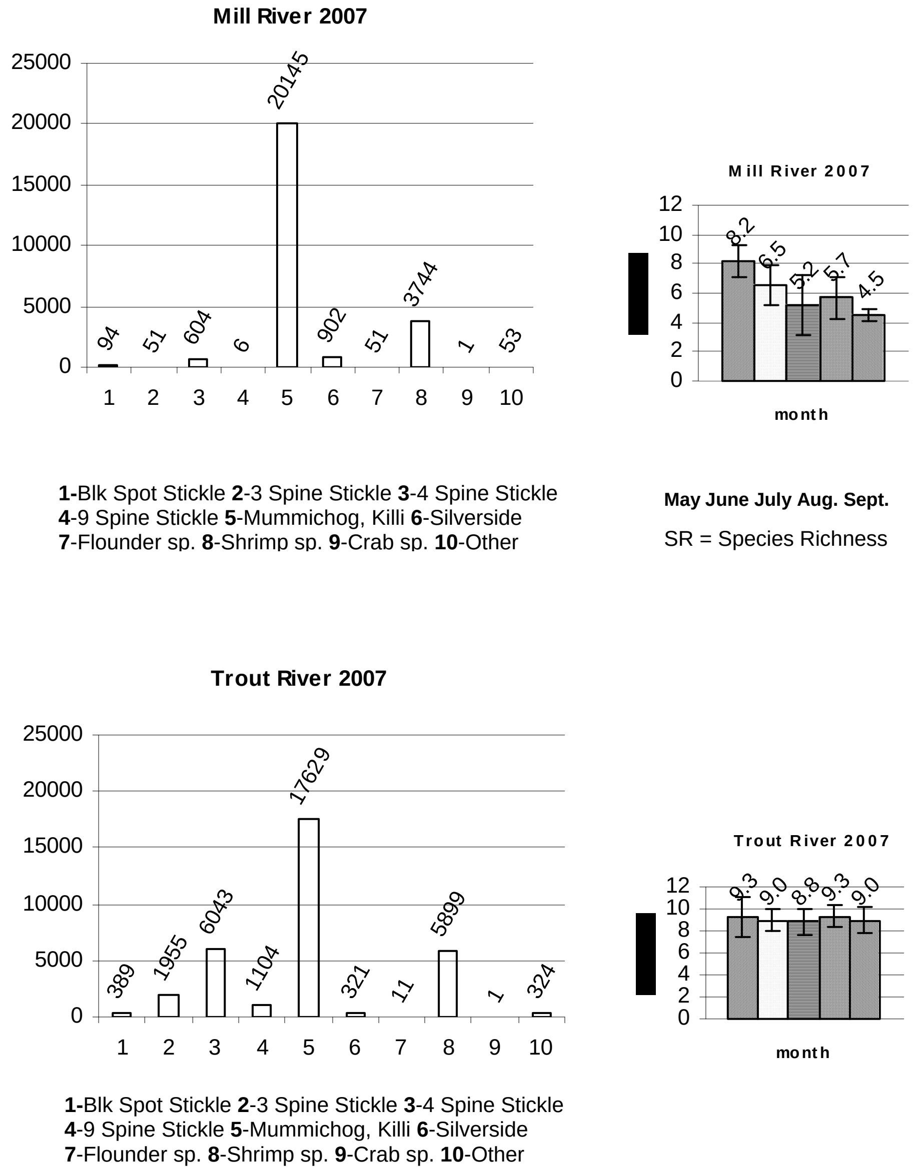


Figure 6. Season totals of adult animals for the 10 most abundant species or species groups in Mill River and Trout River (PEI) sampled over 5 months in 2007. Mean taxon richness (total 6 number of species taxa) is also shown for each month (95 % confidence interval CI) (n = 6 stations per bar).

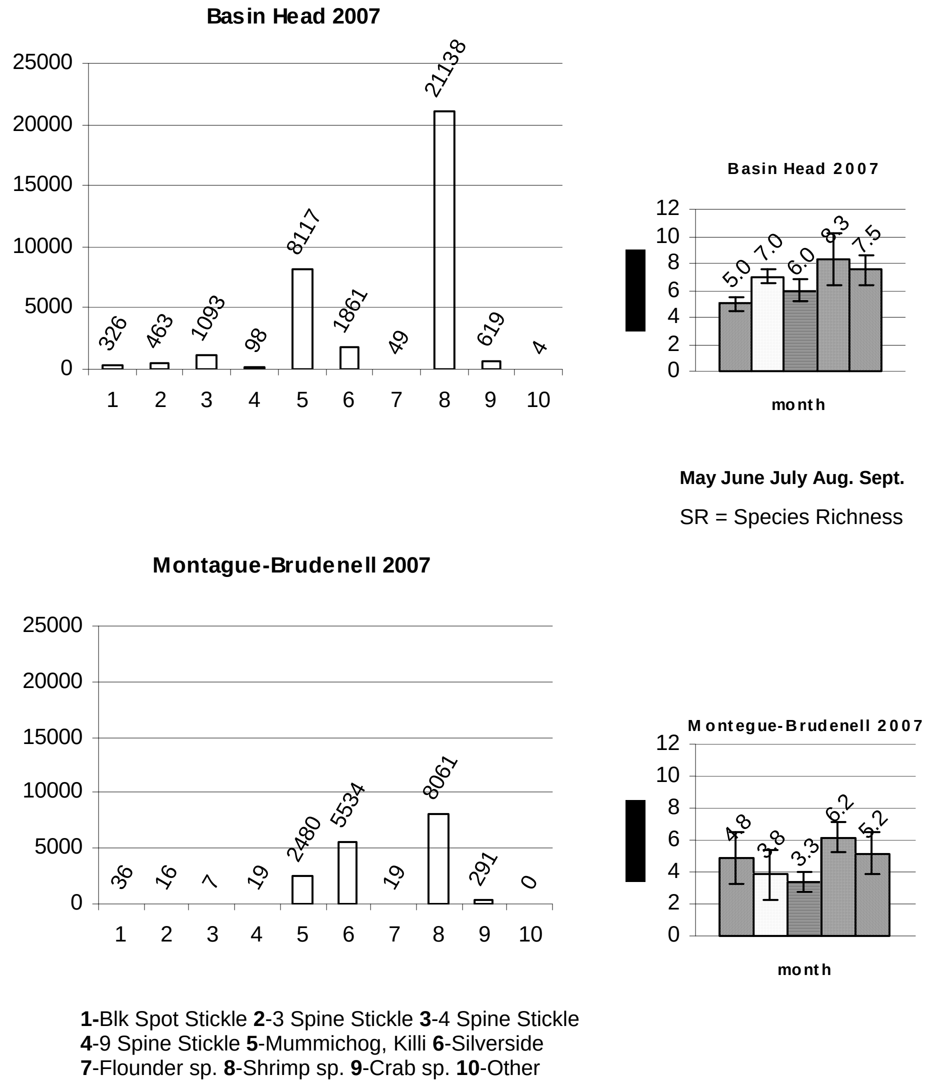


Figure 6A. Season totals of adult animals for the 10 most abundant species or species groups in Basin Head and Montague - Brudenell (PEI) sampled over 5 months in 2007. Mean taxon richness (total 6 number of species taxa) is also shown for each month (95 % confidence interval CI) (n = 6 stations per bar).

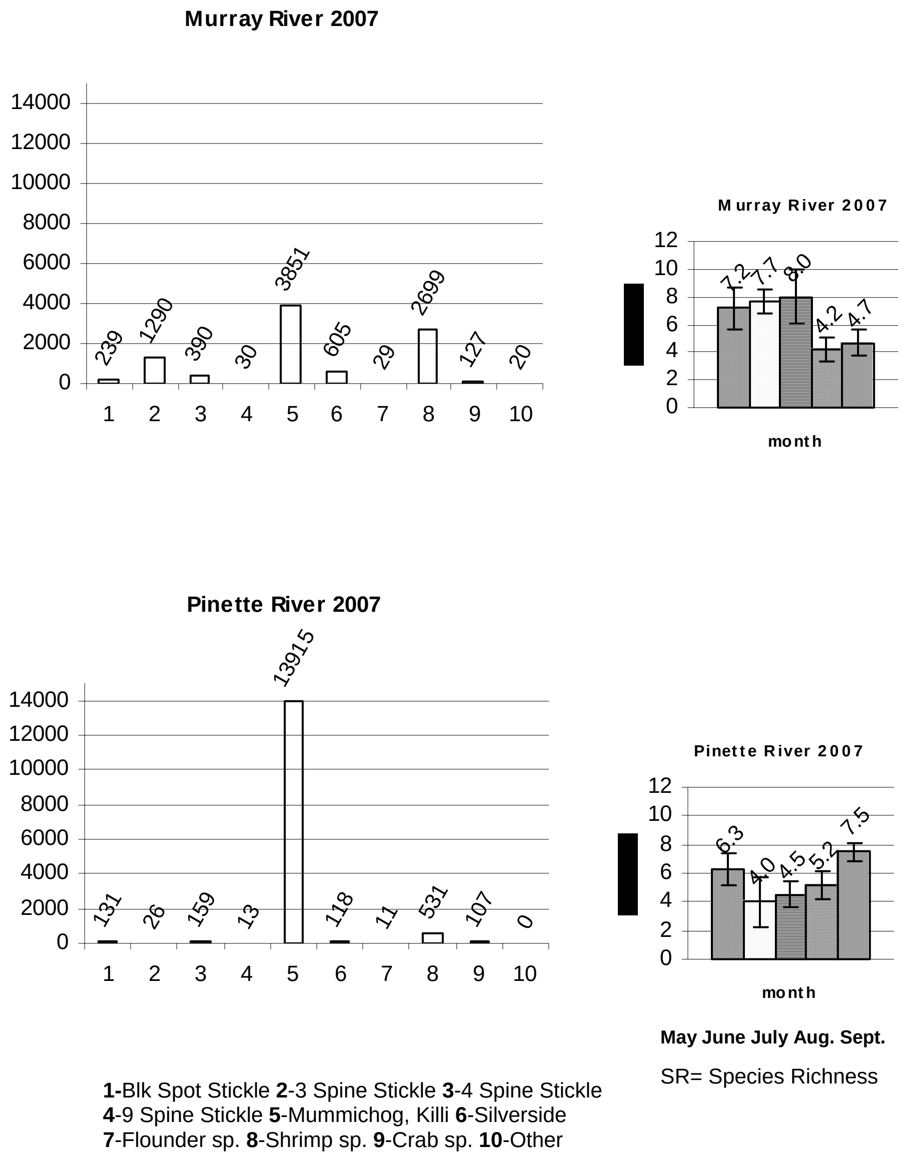


Figure 6B. Season totals of adult animals for the 10 most abundant species or species groups in Murray and Pinette Rivers (PEI) sampled over 5 months in 2007. Mean taxon richness (total 6 number of species taxa) is also shown for each month (95 % confidence interval CI) (n = 6 stations per bar).

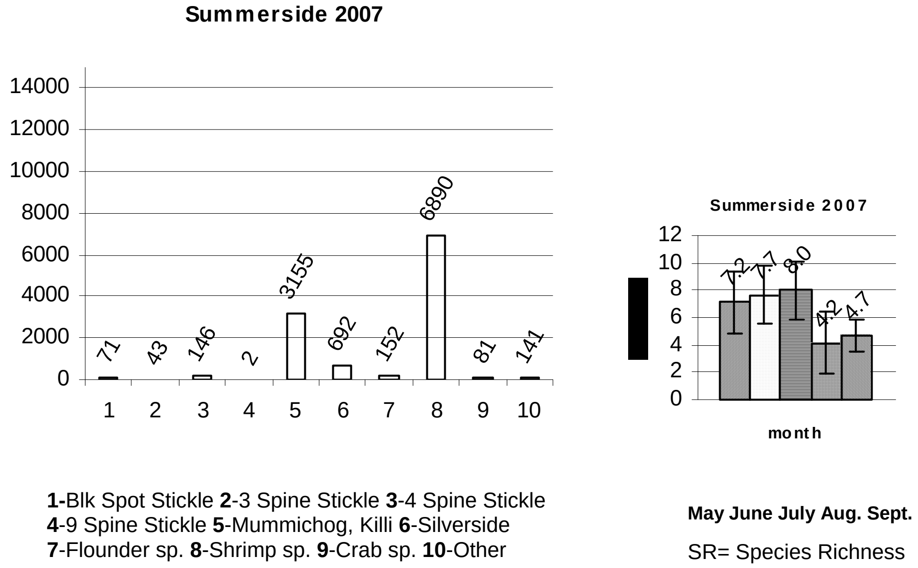


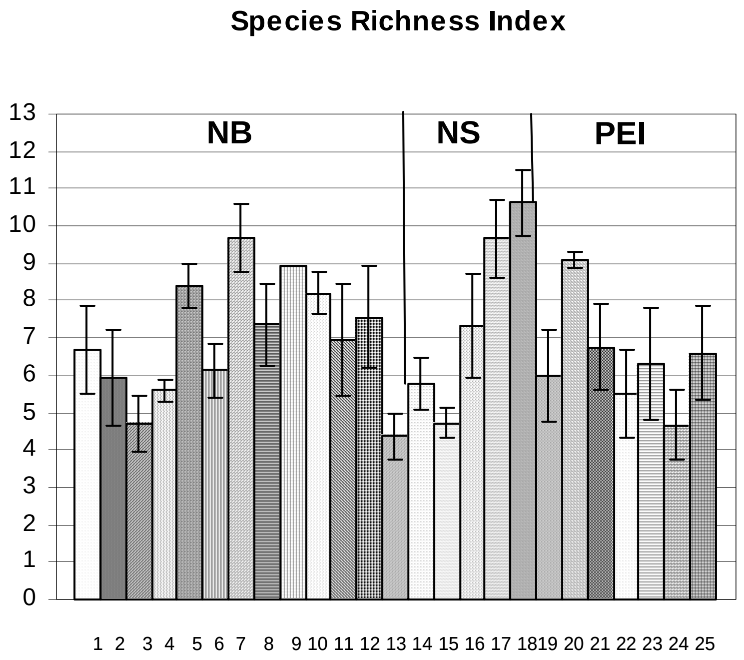
Figure 6C. Season totals of adult animals for the 10 most abundant species or species groups in Bedeque Bay (Summerside) (PEI) sampled over 5 months in 2007. Mean taxon richness (total 6 number of species taxa) is also shown for each month (95 % confidence interval CI) (n = 6 stations per bar).

In Prince Edward Island (Figure 6, 6A & 6B), Basin Head, Trout River and Mill River again maintained their baseline status in 2007. The Southeast Environmental Association (SEA) completed CAMP at the Pinette, Murray and Montague-Brudenell river estuaries to return to full five month baseline status in 2007. The Bedeque Bay Summerside group became the new addition in 2007 and completed the full five months of sampling.

Comparing the seven baseline locations in PEI, *Fundulus* sp. had the highest totals in Mill, Summerside, Trout, Murray and Pinette rivers. *Fundulus* sp. was second and third, respectively, in Basin Head and Montague-Brudenell. Shrimp species were highest in total numbers in Basin Head, Montague-Brudenell and Summerside and second most abundant in Mill, Trout, Murray and Pinette. Overall, the three longest standing baseline sites, Mill River, Trout River and Basin Head show higher total numbers in 2007 than in 2006. The others cannot be compared because of less frequent sampling compared to the previous year.

For the stickleback species, fourspine abundance was highest at all locations except Montague-Brudenell and Murray river locations, similar to what was found in 2006. The threespine stickleback abundance was second highest in Trout River and Basin Head, locations and the blackspotted second at Mill and Pinette while being number one in abundance in Montague-Brudenell. Trout River had a very high number of ninespine sticklebacks compared to all the other locations.

Mean species richness showed no consistent pattern over the months, though higher numbers generally occurred in May except for Basin Head and Montague-Brudenell where higher numbers came later in the sampling season. Species richness comparisons showed a slight drop as the season progressed in Mill River, Murray River and Summerside but remained steady in the other four locations. Once again, Trout River had the higher mean species richness numbers compared to the other locations.



(NB) 1- Caraquet **2**- Shippagan **3**- Lamèque **4**- Tracadie **5**- Tabusintac
6- Miramichi **7**- St. Louis de Kent **8-** Richibucto **9**- Bouctouche **10**- Cocagne
11-Shediac **12**- Scoudouc **13**- Cape Jourimain **(NS) 14**- R. Philip **15**- Pugwash
16- Pictou **17**- Antigonish **18**- Mabou **(PEI) 19-** Mill River **20**-Trout River
21- Basin Head **22-** Pinette R. **23**- Murray R. **24-** Montague-Brudenell R.
25- Summerside

Figure 7. The mean species richness with a confidence interval at ± 95% (n = 5 months for each bar; total species observed at all six stations averaged for each month) for 25 baseline estuary locations in the Gulf of St. Lawrence over a 5 month period

Figure 7 provides an average species richness of all 6 sample stations, for all months for each of the 25 baseline estuaries or coastal shoreline locations. The species richness for all five months was averaged to arrive at the reference value shown in the graph. At this time the combined data provide a baseline reference point. When more information is available, environmental science personnel will examine the successive year data for patterns and trends.

In New Brunswick, St. Louis de Kent had the highest average species richness for the five month sampling period at 9.70 species ± 0.90 (SD), followed by Bouctouche (8.93) and Tabusintac (8.40). The lowest average species richness was in Cape Jourimain at 4.37 species ± 0.60. Trends from 2007 show four locations increased in average species richness;

Caraquet (5.1-6.9), Tabusintac (7.2-8.4), Miramichi (4.7-6.1) and Cocagne (7.4-8.2) while three locations decreased namely Lamèque (5.4-4.7), Richibucto (8.1-7.4) and Cape Jourimain (5.4-4.4). Trends in 2007 shows that the average SR increased from 2006 values in four locations.

In Nova Scotia, Mabou Harbour had the highest species richness value at 10.63 ± 0.87 (SD) followed closely by Antigonish with 9.67 ± 1.04 (SD) both having similar results to 2007 levels. Pugwash was lowest at 4.73 species ± 0.38 (SD). Pictou remained the same as 2007 levels at 7.33 average species richness.

In PEI, Trout River had the highest mean species richness value at 9.10 species ± 0.20 (SD), up from the 2006 value of 8.32. Basin Head and Mill River had almost identical species richness values at 6.80 ± 1.14 (SD) and 6.00 ± 1.24 (SD) respectively in 2007. For the three locations in the southeast sampled as full baseline in 2007 the Montague - Brudenell system has the lowest value at 4.67 species ± 0.94 (SD) similar to the 2006 levels. The other two locations, Pinette and Murray Rivers were slightly higher at 5.5 and 6.3 respectively in 2007, again similar to their levels in 2006.

3.2 Substrate Characteristics

For each site at all locations the percentage of substrate characteristics were recorded each month. The three dominant substrates were mud, sand or rock. This could change depending on the distance sampled from the high water mark. Though some sites would show no change whether the seine was hauled at low or high tide, others easily could. The most often encountered situation would be sand close to shore, mud-sand part way out and mud further out. This is one of the reasons that groups were encouraged to try and sample at similar water depths each month. A multitude of logistic factors contribute to this not always being possible, but for the majority of visits it was. **Table 1** below provides an average of percentage of the four dominant substrate types for each station at every location. The authors refer to a site as primarily of one substrate type if the average (for five months) value determined for the station is greater than 50 percent for the dominant substrate. Keep in mind that groups might describe the station as mostly mud at lower tides as compared to calling it mostly sand at higher tides. As can be seen from the table, in New Brunswick, there are approximately six rocky stations with some sand, 12 primarily muddy stations with some sand, three stations that are half and half, mud and sand and the rest (57) are sandy stations with various combinations of some mud, gravel and/or rock. One of the reasons that sand is the dominant substrate has to do with location, the other with tides. Groups tend to sample the shoreline when the tide is more in than out and locations that have driving access often relates to the public being able to get to locations suitable for recreation. Historically, access roads more often end up at a sandy beach location than at a muddy beach location. Comparing the sites from successive years will provide a general substrate description after all the volunteer categorizations are averaged. This can be best accomplished in the five year summary report,

Table 1. Summary of the dominant bottom sediment type observed for six stations of each estuary location in New Brunswick based on the average of the recorded percentage for the five months of sampling in 2007

Cara		avg %		
stn	sand	grav	mud	rock
1	43	16	0	41
2	97	3	0	0
3	95	5	0	0
4	67	19	0	14
5	31	6	44	20
6	55	31	2	12

Lam		avg %		
stn	sand	grav	mud	rock
1	80	1	6	13
2	61	7	26	6
3	74	5	13	8
4	60	8	24	8
5	72	8	15	5
6	55	8	13	24

Ship		avg %		
stn	sand	grav	mud	rock
1	80	3	15	2
2	61	5	30	4
3	40	21	7	32
4	70	6	16	8
5	58	12	26	4
6	57	11	18	14

Trac		avg %		
stn	sand	grav	mud	rock
1	76	6	17	1
2	80	2	18	0
3	69	8	22	1
4	76	4	19	1
5	73	9	18	0
6	58	9	32	1

Tabus		avg %		
stn	sand	grav	mud	rock
1	62	13	13	12
2	78	3	19	0
3	56	2	42	0
4	48	12	4	36
5	50	0	50	0
6	60	0	40	0

Miram		avg %		
stn	sand	grav	mud	rock
1	89	6	3	2
2	50	0	30	20
3	80	6	0	14
4	43	10	41	6
5	89	3	0	8
6	21	7	0	72

St.Lou		avg %		
stn	sand	grav	mud	rock
1	80	0	20	0
2	73	2	24	1
3	27	18	48	7
4	35	7	57	1
5	34	20	44	2
6	32	11	54	3

Rich		avg %		
stn	sand	grav	mud	rock
1	51	26	13	10
2	62	23	7	8
3	54	4	1	51
4	60	9	9	22
5	27	17	0	56
6	68	21	4	7

Bouct		avg %		
stn	sand	grav	mud	rock
1	56	4	36	4
2	38	2	48	12
3	90	1	9	0
4	64	6	20	10
5	34	14	46	6
6	42	8	44	6

Cocag		avg %		
stn	sand	grav	mud	rock
1	87	5	2	6
2	29	5	63	3
3	77	13	5	5
4	69	8	14	9
5	63	7	16	14
6	75	18	1	6

Shed		avg %		
stn	sand	grav	mud	rock
1	25	2	68	5
2	69	1	30	0
3	65	0	34	1
4	60	6	34	0
5	48	8	24	20
6	68	4	28	0

Scou		avg %		
stn	sand	grav	mud	rock
1	54	6	40	0
2	60	3	37	0
3	96	1	3	0
4	49	1	50	0
5	73	3	22	2
6	68	0	32	0

Jouri		avg %		
stn	sand	grav	mud	rock
1	95	4	0	1
2	43	4	47	6
3	88	5	0	7
4	76	10	0	14
5	71	2	0	27
6	59	21	0	20

In Prince Edward Island, as can be seen from **Table 2**, there are no stations where rock is dominant, though three in both Murray and Summerside and one in Trout have a rock base covered with sand. There are seven primarily muddy stations with some sand (two in each of Trout and Basin Head, one in Summerside), six stations that are approximately half and half, mud and sand (two in Mill and Pinette, one in each of Trout and Basin Head) and the rest (31) are sandy stations with various combinations of some mud, gravel and/or rock. As before, above 50 percent for the average of the five monthly observations by volunteers is the value used to say what kind of substrate is dominant, or what combination dominates.

In Nova Scotia (also **Table 2**), there are no rocky stations (one is close at 41% in Philip), three that are mostly gravel with some rock and sand (two in Mabou, one in Antigonish), six primarily muddy stations (three in Philip, two in Pugwash, one in Antigonish), three sand/gravel, eight stations that are approximately half and half, mud/sand or mud/gravel and the rest (nine) are sandy stations with various combinations of some mud, gravel and/or rock.

Table 2. Summary of the dominant bottom sediment type observed for six stations of each estuary location in Nova Scotia and Prince Edward Island based on the average of the recorded percentage for the five months of sampling in 2007

PEI

Mill	**avg %**			
stn	**sand**	**grav**	**mud**	**rock**
1	59	5	12	4
2	95	1	4	0
3	86	4	7	3
4	77	3	20	0
5	59	0	40	1
6	50	0	50	0

Sum	**avg %**			
stn	**sand**	**grav**	**mud**	**rock**
1	88	4	3	5
2	50	1	14	35
3	56	0	8	36
4	32	2	64	4
5	67	9	8	16
6	47	7	16	30

Trou	**avg %**			
stn	**sand**	**grav**	**mud**	**rock**
1	30	2	68	0
2	34	0	66	0
3	44	4	52	0
4	65	0	35	0
5	73	0	3	24
6	76	6	18	0

Pine	**avg %**			
stn	**sand**	**grav**	**mud**	**rock**
1	89	3	3	5
2	68	0	32	0
3	58	0	32	10
4	78	4	10	8
5	89	9	2	0
6	72	3	24	1

Murr	**avg %**			
stn	**sand**	**grav**	**mud**	**rock**
1	70	2	0	28
2	69	7	0	24
3	60	7	0	24
4	71	16	0	13
5	75	6	9	10
6	75	0	19	6

Mont	**avg %**			
stn	**sand**	**grav**	**mud**	**rock**
1	80	9	0	11
2	98	0	0	2
3	73	5	18	4
4	81	0	19	0
5	81	6	1	12
6	88	0	10	2

BasH	**avg %**			
stn	**sand**	**grav**	**mud**	**rock**
1	63	2	35	0
2	86	0	14	0
3	31	0	69	0
4	60	0	40	0
5	38	2	56	4
6	19	0	81	0

NS

Phil	**avg %**			
stn	**sand**	**grav**	**mud**	**rock**
1	34	23	2	41
2	45	38	1	16
3	29	14	54	3
4	62	9	26	3
5	23	23	54	0
6	30	6	60	4

Pugw	**avg %**			
stn	**sand**	**grav**	**mud**	**rock**
1	2	2	96	0
2	16	0	94	0
3	83	11	0	6
4	62	18	0	20
5	96	0	0	4
6	74	10	0	16

Pict	**avg %**			
stn	**sand**	**grav**	**mud**	**rock**
1	21	26	36	17
2	36	26	36	2
3	5	45	33	17
4	66	22	8	4
5	95	5	0	0
6	60	18	12	10

Anti	**avg %**			
stn	**sand**	**grav**	**mud**	**rock**
1	7	34	42	17
2	12	40	36	12
3	48	50	0	2
4	0	0	100	0
5	39	22	16	23
6	71	16	8	5

Mab	**avg %**			
stn	**sand**	**grav**	**mud**	**rock**
1	26	50	33	11
2	14	44	24	18
3	12	38	43	7
4	7	46	27	20
5	48	34	12	6
6	22	50	16	12

3.3 Substrate Composition

In September, one sediment sample was taken at every sample station at all locations. Each group used a 165 cm blade trowel to dig into the sediment. Depth was restricted to elbow depth in the water. The samples were analyzed by a Coop student in our Gulf Region laboratory in the spring of 2008. The three dominant sediment types were sand and mud or a combination of both. The sediment analysis was completed in the fall by lab technicians at DFO Gulf region. A more detailed description of the procedure is included in Weldon et al. 2005.

Table 3. Summary of average % organic content (± S.D.), % moisture content (± S.D.), and mean grain size (MGS) for all the baseline locations (n = 6).

	% Moisture	% Organic	mean Grain Size MGS (mm)
N.B.	**September**		
Caraquet	**21.18 ± 3.78**	**0.99 ± 0.22**	**0.32 ± 0.16**
Lamèque	**22.47 ± 2.59**	**1.53 ± 0.17**	**0.61 ± 0.68**
Shippagan	**20.53 ± 2.49**	**0.96 ± 0.25**	**0.47 ± 0.12**
Tracadie	**25.84 ± 1.33**	**1.70 ± 0.56**	**0.30 ± 0.09**
Tabusintac	**22.72 ± 4.06**	**1.56 ± 0.34**	**0.62 ± 0.55**
Miramichi	**30.67 ± 14.62**	**4.35 ± 6.56**	**0.28 ± 0.12**
St. Louis de Kent	**23.41 ± 2.70**	**1.43 ± 0.54**	**0.34 ± 0.26**
Richibucto	**22.10 ± 1.41**	**1.27 ± 0.37**	**0.56 ± 0.51**
Bouctouche	**27.81 ± 9.05**	**3.18 ± 1.77**	**0.36 ± 0.23**
Cocagne	**21.13 ± 2.78**	**1.15 ± 0.65**	**0.47 ± 0.27**
Shediac	**29.51 ± 10.82**	**2.12 ± 1.54**	**0.37 ± 0.08**
Scoudouc	**22.48 ± 1.58**	**1.21 ± 0.29**	**0.25 ± 0.04**
Jourimain	**23.97 ± 2.75**	**0.55 ± 0.19**	**0.54 ± 1.71**
average for NB	**24.14 ± 4.6**	**1.69 ± 0.55**	**0.42 ± 0.37**
N.S.	**% Moist**	**% Organic**	**MGS**
Philip	**20.77 ± 3.85**	**0.28 ± 0.12**	**0.61 ± 0.72**
Pugwash	**30.98 ± 27.85**	**9.25 ± 20.15**	**0.79 ± 0.67**
Pictou	**22.40 ± 12.92**	**2.33 ± 2.70**	**1.00 ± 0.83**
Antigonish	**22.18 ± 16.61**	**2.67 ± 4.07**	**1.75 ± 0.75**
Mabou	**23.69 ± 5.75**	**0.87 ± 0.60**	**0.61 ± 0.72**
average for NS	**24.00 ± 13.40**	**3.08 ± 5.53**	**0.95 ± 0.74**
PEI	**% Moist**	**% Organic**	**MGS**
Mill River	**21.00 ± 4.07**	**0.68 ± 0.26**	**0.50 ± 0.60**
Trout River	**23.75 ± 3.54**	**1.16 ± 0.74**	**0.53 ± 0.73**
Summerside	**18.70 ± 2.73**	**0.53 ± 0.12**	**0.35 ± 0.14**
Basin Head	**30.09 ± 16.55**	**3.18 ± 5.60**	**0.31 ± 0.06**
Murray	**15.65 ± 5.40**	**0.74 ± 0.22**	**1.17 ± 0.73**
Mont.- Brudenell	**21.27 ± 4.06**	**0.60 ± 0.27**	**0.59 ± 0.52**
Pinette	**23.98 ± 5.59**	**1.51 ± 1.47**	**0.29 ± 0.07**
average for PEI	**22.06 ± 5.99**	**1.20 ± 1.24**	**0.53 ± 0.41**

In New Brunswick, the Miramichi samples had the highest average value for percent moisture content at 30.67 with Shediac close behind at 29.51, though all sites fell between 20.53 and

30.67. In New Brunswick, percent moisture content averaged 24.1% over all 13 locations. This is an increase of almost two percentage points from last year at 22.3 %.

Miramichi has the highest percent organic content (4.35) compared to the next nearest in Bouctouche (3.18). Every location had lower values for organic content than in 2006. The average for percent organic was 1.7% in 2007, down from the 5.4% in 2006 and similar to the 2.0% in 2005. The average mean grain size for the 13 locations was 0.42 which corresponds to fine sand category (Weldon et al. 2005) compared to 0.47 in 2006 for 12 locations.

In Nova Scotia, Pugwash had the highest percent moisture content at 30.98 compared to the lowest at 20.77. Percent moisture content averaged 24.0% over all five locations. This is an increase from the last year average at 20.8 %. Pugwash also had the highest percent organic content at 9.25. The average for the five locations for the percent organic content was 3.1% up from the 2.3% in 2006. The average mean grain size for the five locations was 0.95 which corresponds to coarse to medium sand compared to 1.3 in 2006 for two locations (other two not available).

In Prince Edward Island, the Basin Head samples had the highest average value for percent moisture content at 30.09 with the lowest being 15.65. Percent moisture content averaged 22.1% over all seven locations. This is a slight increase from last year at 21.3 %. Basin Head also had the highest percent organic content at 3.18. The average for seven locations for percent organic was 1.2% in 2007, down from the 1.6% in 2006. The average mean grain size for the seven locations was 0.53 which corresponds to medium sand compared to 0.28 in 2006 for six locations.

One general observation is that mud bottoms with characteristic small mean grain size had higher percentages of both organic matter and moisture. The opposite patterns exist for sandy bottom locations. Overall, all three provinces show increases for average percent moisture, two show decreases for average total organic and two show decreases for average total mean grain size.

Each time in September a sediment sample is taken in approximately the same place, but keep in mind the effect stage of tide has on the sampling. As referred to repeatedly, mud can be taken at lower tide and sand at a higher tide. Averaging the results over the five years will provide an accurate averaged description of the sites at each location.

3.4 Macrophyte Cover

Percent vegetation cover was estimated using the sampling grid as described in Weldon et al. (2005). A modification to the standardized method for evaluating macrophyte coverage will reduce variability among volunteer samplers. In 2007, each NGO was introduced to the use of a one to five number scale to provide percentage vegetative cover in each or the four sections of the standard quadrat. Number "0" means zero percent vegetation present. Number "5" means 100% vegetation cover. Numbers one, two, three and four represent 0 - <25%, 25 - <50% and 50 - <75%, 75 - < 100% respectively.

For **Tables 4** and **5**, the legend or code was developed to indicate what plant material was present in the quadrat. This enables the individual NGO's to look at the stations and see patterns or trends that may be taking place over the five months of sampling. All together there is data for 12 squares per station (three throws of the quadrat times four sub-quadrats) which was averaged across the 12 sub-squares. This table only describes more than or less than 50% of the squares having vegetation. It does not summarize what percentage of vegetation cover is seen in the square, only that there was vegetation present. Exact percentages can be found in data sheets for the sites, and this may be presented in the summary report to be available in 2009. The table uses the categories from the newly refined vegetation data sheet as reference to what dominant vegetation was present. The categories from the data sheet include the following:

Seed Plants includes Eelgrass - *Zostera marina* and Widgeon (Ditch) Grass - *Ruppia maritime* represented by the letter "S or s".

Green Seaweeds includes Sea Lettuce - *Ulva lactuca*, Hollow Green Weed - *Enteromorpha sp.* and Green Fleece - *Codium fragile* represented by the letter "G or g".

Brown Seaweeds includes the Rockweeds - *Fucus sp.*, Kelps - *Laminaria sp.*, Tangleweeds and Knotted Wrack - *Ascophyllum nodosum* represented by the letters "B or b".

The less plentiful others were represented by "O or o". These would include representatives from the following, listed in order of frequent to less frequently encountered: the various green, brown and red filamentous seaweed species and the red seaweed - Irish Moss - *Condrus crispus*. Some refinement of these categories will occur in 2008.

The following legend should be used to interpret what was seen at the site for each location for each of the five months the site was sampled.

S = Seed plants mostly eelgrass; capital "S" means six or more quadrates with vegetation, "s" is less than six of the 12 small sub-quadrate had vegetation

G = Green algae mostly Ulva; capital "G" means six or more, "g" is less than six of the 12 small sub-quadrates had vegetation

B = Brown algae mostly Fucus; capital "B" means six or more, "b" is less than six of the 12 small sub-quadrates had vegetation

O = Others which could include filamentous species or red algae; capital "O" means six or more, "o" is less than six of the 12 small sub-quadrates had vegetation

The first letter indicates the more abundant species recorded on the day of sampling. A zero "0" means no vegetation was recorded (note the subtle difference of the letter "O" and the number "0"). If wind and wave action made viewing impossible, the not available "n/a" was used. This was also used for the rare case where volunteers forgot to record anything during their visit.

Table 4. Composition of the vegetation profile showing presence in the quadrat from six sample sites at all locations in New Brunswick for the five months in 2007.

	Cara			**stn**		
mth	**1**	**2**	**3**	**4**	**5**	**6**
May	b-s	0	s	0	n/a	g
June	B-G	0	s	G	n/a	G-b
July	B	0	S-o	o	0	s
Aug.	B	0	s	n/a	0	G-b
Sept.	B	n/a	n/a	n/a	n/a	G-b

	Lamè			**stn**		
mth	**1**	**2**	**3**	**4**	**5**	**6**
May	s-g	n/a	G-o	n/a	b-g	b
June	S-G	G-o	G	S	n/a	n/a
July	0	g	G-b	0	G-b	S-G
Aug.	G-s	S-g	G-b	G	S-G	S-G
Sept.	S-g	S-g	G	G	Sgb	s-G

	Shipp			**stn**		
mth	**1**	**2**	**3**	**4**	**5**	**6**
May	0	0	b	0	n/a	0
June	O	S	G	G	0	S
July	S	s	S	S	sgb	S
Aug.	S-O	S-G	s-g	0	G	s-G
Sept.	G	G	G	0	g	g

	Trac					
mth	**1**	**2**	**3**	**4**	**5**	**6**
May	0	0	s	0	s	0
June	s	s	S	S	S	S
July	S	S	S	S	S	S
Aug.	S	S-g	S	S	S	S
Sept.	S	0	S	s	S	s

	Tabu					
mth	**1**	**2**	**3**	**4**	**5**	**6**
May	s-g	0	n/a	n/a	n/a	0
June	o	0	s-o	n/a	n/a	g
July	0	0	s	0	S	0
Aug.	0	0	0	0	0	0
Sept.	0	0	S	0	S	0

	Mira					
mth	**1**	**2**	**3**	**4**	**5**	**6**
May	0	0	0	n/a	n/a	s-o
June	n/a	g	o	g	n/a	n/a
July	0	0	0	n/a	sgb	0
Aug.	o	o	0	n/a	0	0
Sept.	0	b	n/a	n/a	n/a	s-o

	St.L					
mth	**1**	**2**	**3**	**4**	**5**	**6**
May	S	n/a	0	0	s	s
June	S	0	S	s	s-g	S
July	S	S	S	s	Sgo	S
Aug.	0	s	s	S	Sgo	S
Sept.	s	S	0	s	0	s

	Rich					
mth	**1**	**2**	**3**	**4**	**5**	**6**
May	0	0	0	0	0	S
June	s	0	0	0	0	0
July	s	b	b	0	b	S-b
Aug.	0	0	0	0	0	S
Sept.	0	S	0	s	g	0

	Bouct					
mth	**1**	**2**	**3**	**4**	**5**	**6**
May	n/a	n/a	0	n/a	n/a	g-o
June	0	n/a	g	G	G	G-o
July	0	n/a	g-s	g	0	n/a
Aug.	0	n/a	0	G	n/a	S-G
Sept.	g-b	g	n/a	gbo	G-o	G-s

	Coca					
mth	**1**	**2**	**3**	**4**	**5**	**6**
May	o	n/a	0	n/a	n/a	n/a
June	sgo	s-o	S-g	n/a	G	gos
July	S-O	S	0	S-g	S-G	sgo
Aug.	0	G	S-O	n/a	n/a	n/a
Sept.	s	S-g	s-G	s-G	n/a	S-g

	Shed					
mth	**1**	**2**	**3**	**4**	**5**	**6**
May	G-O	O	s	o	S	S
June	s-g	sgo	0	g-o	g-b	s
July	S-b	0	0	g	0	0
Aug.	n/a	Sgo	n/a	G	g-O	SGO
Sept.	s	S-g	S	G-s	O-s	SgO

	Scou					
mth	**1**	**2**	**3**	**4**	**5**	**6**
May	0	n/a	b	o	0	n/a
June	0	0	s-o	0	O	S-O
July	0	sgo	s	G-s	SGO	S-O
Aug.	0	g-o	0	0	0	g
Sept.	g	g	0	g	g-s	0

	Jouri					
mth	**1**	**2**	**3**	**4**	**5**	**6**
May	0	0	0	0	0	0
June	n/a	n/a	0	o	s	B
July	o	o	0	0	o	o
Aug.	0	o	0	0	o	b
Sept.	0	0	0	0	0	0

Table 5. Composition of the vegetation profile showing presence in the quadrat from six sample sites at all locations in Prince Edward Island and Nova Scotia for the five months in 2007.

PEI

	Mill			stn		
mth	1	2	3	4	5	6
May	g	s-g	S	S	G-s	G
June	G	S-G	O	o	G	g
July	n/a	G	G	g	G	G
Aug.	G	G	0	0	G	G
Sept.	G	0	g	G	g	G

	Sum			stn		
mth	1	2	3	4	5	6
May	b-o	b	b	0	0	0
June	n/a	n/a	0	g	g	G
July	n/a	G	s	0	g	g
Aug.	g	G	g	G	G	G
Sept.	S-g	n/a	s	g	g	g

	Trou			stn		
mth	1	2	3	4	5	6
May	G	g	S	S	sgo	g
June	n/a	n/a	S	S-g	sgo	s-G
July	S-o	n/a	S	S	S-O	S-g
Aug.	Sgo	0	S	S-G	Sbo	G
Sept.	s-g	G	S	s-G	sgb	G

	Basin					
mth	1	2	3	4	5	6
May	s-g	0	s-g	0	s	s
June	S	S	S	n/a	s	G
July	0	0	s	S	g	G
Aug.	S	S	0	g	s	G
Sept.	0	0	0	0	0	sgo

	Mon -	Bru				
mth	1	2	3	4	5	6
May	0	0	0	0	0	0
June	0	0	0	0	0	0
July	0	0	0	0	0	0
Aug.	0	0	0	0	0	0
Sept.	s	S	0	0	S	s

	Murr					
mth	1	2	3	4	5	6
May	0	0	0	0	g-b	0
June	0	0	n/a	g	0	n/a
July	n/a	0	S	s	S	S
Aug.	s	s	0	s	s	0
Sept.	0	s	0	0	s	0

	Pine					
mth	1	2	3	4	5	6
May	0	0	n/a	n/a	s	S
June	S	0	0	0	0	0
July	S	S	0	0	0	s
Aug.	S	S-g	s-o	o	s	S
Sept.	s	S	s-o	0	s	S

NS

	Phil					
mth	1	2	3	4	5	6
May		no		sample		
June	B	0	S	s-g	o	0
July	B	b	S	g	O	0
Aug.	B	0	S	S	O	0
Sept.	B	s-o	O	O	b-O	O

	Pugw					
mth	1	2	3	4	5	6
May	S	0	b-o	b	0	0
June	S	b	0	O	0	n/a
July	0	S	b	b	0	s-b
Aug.	O	S	n/a	n/a	b	n/a
Sept.	O	O	0	0	0	0

	Pict					
mth	1	2	3	4	5	6
May	G	G-O	G	0	0	b
June	n/a	G-O	gbo	0	n/a	b
July	o	n/a	g	0	0	b
Aug.	o	0	g-o	0	0	0
Sept.	s-g	s	g	0	n/a	0

	Anti					
mth	1	2	3	4	5	6
May	g	S-g	o	O	O	O
June	S	n/a	g-o	0	b-O	0
July	S	s-G	b-o	0	0	b
Aug.	O	B-o	s	0	0	0
Sept.	S	B-o	b-o	0	o	0

	Mabo					
mth	1	2	3	4	5	6
May	g	B	B	g-b	G-B	B
June	S	S	s	B	G-B	B
July	S	S-b	s	s-b	G-b	B
Aug.	S	S-b	S	sgb	S	B
Sept.	S	S	S	s-B	S	B

When considering all locations from all three provinces, the dominant vegetation present was the submerged eelgrass beds. When present, the eelgrass bed was located in the outer 1/5th or just beyond the sample sweep area. Generally, upper estuary sites in the river system contain more eelgrass beds than the outer portion that is closer to the mouth or associated with a beach area. When a vegetation bed was present, the number of fish and invertebrates collected was usually higher as more habitats for the pelagic species were available. An increase in vegetation was most noticeable where mud bottom substrates supported eelgrass vegetation. When sampling at higher tides, the vegetation bed was often not swept by the beach seine. A continuing concern to NGO volunteers was the accumulation of unattached material in the beach seine at a sample site. Large collections of floating algae such as sea lettuce and eelgrass could hinder the effectiveness of collections by smothering fish before they could be counted and released unharmed. Since it was impossible to sweep a net through certain sites in mid-summer, due to the large volumes of un-attached algae, some station relocations were necessary. These stations were moved, but for as short a distance as possible, so a level of comparability was maintained. This year, the moving of sites was minimal, but potential changes are evaluated at the end of each season after consultation with area office coordinators.

Sites with larger rocks are likely to include species of rockweed (usually *Fucus vesiculosus*). Other abundant macrophyte vegetation included sea lettuce (*Ulva lactuca*) and filamentous green algae (*Cladophora sp*.). In northern NB and at a few NS sites hollow green weed (*Enteromorpha intestinalis*) was noticeable but not abundant. On many occasions, NGO volunteers have found floating unattached pieces of kelp species (*Laminaria*) and the invasive green fleece (*Codium fragile*) in the net sweep. These species are usually attached by a holdfast in deeper water. Vegetation type is important as it defines the variety of available habitats. Preference of YOY pelagics and juvenile crustaceans for protective cover from predators usually translates into greater numbers when vegetation is present.

These vegetation cover indicators fluctuated throughout the season and do show variation from year to year. Percent vegetation cover using the one – five number scale is a qualitative measure because some is attached (eelgrass and rockweed) and some is floating as was observed with sea lettuce. The spring observations have lower values for vegetation but as a season progressed, the vegetation present generally increased.

The quadrat method described in Weldon et al. (2005) is effective when the water has not been stirred up. In 2006, it was suggested that the quadrat be thrown parallel to the shore at a mid depth from shore, usually at a distance where the bottom could still be seen. This protocol was followed in 2007 as well. The NGO's were instructed to make a detailed record of the characteristics of the bottom profile for each station each month so comparisons could be made as the season progressed. Characteristic broken off wash up of all plant material in the shoreline berm was also recorded. Over time an averaging of the results of the site for every station will produce the best possible picture of the vegetation profile. Also, the changes from year to year, though often subtle can be determined. But these observations can still be complicated by a variety of factors. The most obvious would be the results recorded can change depending on what stage of tide the site was visited, so it becomes important to try and visit the site at the same tide stage each month and hopefully carry this pattern over to successive years.

3.5 Physical Measures

At each sample location and at every station, three physical parameters were measured on each occasion. Readings for water temperature, salinity and dissolved oxygen provide a monthly snapshot of conditions at each site (**Tables 6 - 8**).

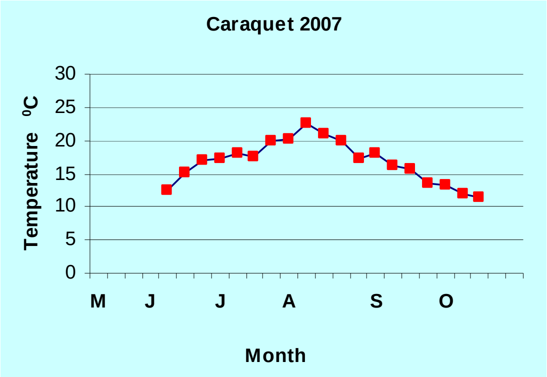
Table 6. Average monthly temperature (^{0}C ± S.D.) per site for the 2007 season (n = 6). (NA = not available)

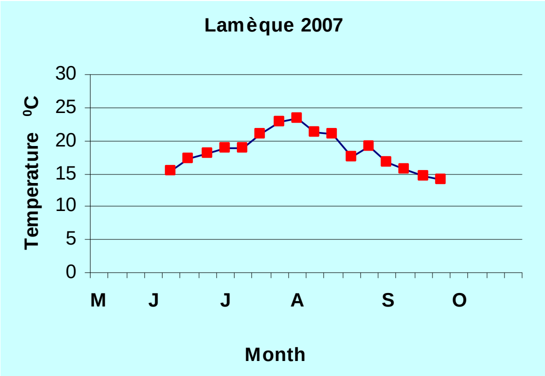
	Temp ^{0}C ± S.D.				
N.B.	**May**	**June**	**July**	**Aug.**	**Sept.**
Caraquet	**14.1 ± 2.6**	**20.3 ± 2.4**	**26.3 ± 3.4**	**21.4 ± 2.7**	**12.4 ± 1.4**
Lamèque	**10.9 ± 0.6**	**21.0 ± 0.6**	**22.8 ± 0.2**	**18.8 ± 0.9**	**11.6 ± 0.4**
Shippagan	**12.3 ± 1.2**	**22.1 ± 1.3**	**28.4 ± 1.5**	**24.8 ± 2.1**	**12.4 ± 0.5**
Tracadie	**12.7 ± 1.4**	**18.5 ± 1.1**	**25.0 ± 0.8**	**20.1 ± 2.3**	**12.9 ± 1.4**
Tabusintac	**14.1 ± 0.8**	**18.1 ± 0.7**	**24.8 ± 0.3**	**19.4 ± 0.6**	**14.0 ± 1.0**
Miramichi	**13.7 ± 1.2**	**19.3 ± 1.0**	**21.4 ± 0.8**	**20.0 ± 1.4**	**16.4 ± 0.9**
St Louis	**14.9 ± 1.2**	**20.7 ± 1.2**	**26.3 ± 1.9**	**21.5 ± 0.7**	**16.9 ± 0.6**
Richibucto	**11.1 ± 0.6**	**18.2 ± 0.9**	**23.1 ± 1.3**	**20.0 ± 0.7**	**15.7 ± 0.7**
Bouctouche	**13.9 ± 2.3**	**19.0 ± 0.1**	**22.4 ± 0.6**	**19.8 ± 2.4**	**19.5 ± 3.5**
Cocagne	**12.1 ± 1.4**	**22.5 ± 2.7**	**25.8 ± 1.4**	**17.4 ± 1.1**	**16.7 ± 1.2**
Shediac	**17.2 ± 2.0**	**21.1 ± 1.3**	**25.5 ± 0.9**	**20.6 ± 1.7**	**14.7 ± 1.3**
Scoudouc	**15.8 ± 0.9**	**18.7 ± 1.9**	**23.2 ± 1.4**	**21.9 ± 0.4**	**18.4 ± 0.3**
Jourimain	**13.3 ± 1.8**	**19.0 ± 1.5**	**22.2 ± 2.2**	**20.5 ± 1.3**	**16.9 ± 0.7**
Average NB	**12.1 ± 1.4**	**19.9 ± 1.3**	**24.4 ± 1.3**	**20.5 ± 1.4**	**15.3 ± 1.1**
N.S.					
Philip	**N/A**	**19.7 ± 0.6**	**24.1 ± 0.7**	**22.1 ± 1.1**	**16.3 ± 1.2**
Pugwash	**10.4 ± 0.8**	**17.7 ± 0.2**	**24.1 ± 2.1**	**20.7 ± 0.4**	**19.6 ± 2.6**
Pictou	**12.9 ± 2.2**	**18.4 ± 2.4**	**25.2 ± 3.3**	**22.4 ± 1.4**	**18.7 ± 2.5**
Antigonish	**13.1 ± 1.7**	**17.7 ± 1.2**	**24.3 ± 1.4**	**22.0 ± 1.2**	**15.7 ± 1.1**
Mabou	**9.6 ± 0.6**	**17.2 ± 1.3**	**24.2 ± 1.1**	**18.5 ± 1.5**	**17.6 ± 1.1**
Average NS	**11.5 ± 1.3**	**18.1 ± 1.1**	**24.4 ± 1.7**	**21.1 ± 1.1**	**17.6 ± 1.7**
P.E.I.					
Mill River	**14.8 ± 1.5**	**19.2 ± 1.0**	**25.1 ± 0.5**	**21.9 ± 1.0**	**17.0 ± 0.4**
Trout River	**15.1 ± 3.1**	**20.2 ± 2.3**	**25.1 ± 1.3**	**20.2 ± 1.3**	**16.8 ± 1.3**
Summerside	**10.7 ± 1.0**	**18.7 ± 1.0**	**19.2 ± 1.2**	**21.6 ± 1.2**	**18.7 ± 2.5**
Basin Head	**8.2 ± 0.4**	**18.7 ± 0.9**	**22.2 ± 0.9**	**20.4 ± 2.6**	**12.9 ± 0.2**
Mont.-Brudenell	**9.7 ± 2.3**	**14.9 ± 0.4**	**18.2 ± 0.9**	**20.5 ± 0.6**	**17.8 ± 0.2**
Murray	**10.6 ± 0.8**	**14.5 ± 0.9**	**19.8 ± 1.3**	**20.9 ± 0.8**	**18.9 ± 0.8**
Pinette	**11.9 ± 0.3**	**13.1 ± 0.6**	**19.8 ± 0.8**	**21.5 ± 0.3**	**19.0 ± 2.8**
Average PEI	**11.6 ± 1.3**	**17.0 ± 1.0**	**21.3 ± 1.0**	**21.0 ± 1.1**	**17.3 ± 1.2**

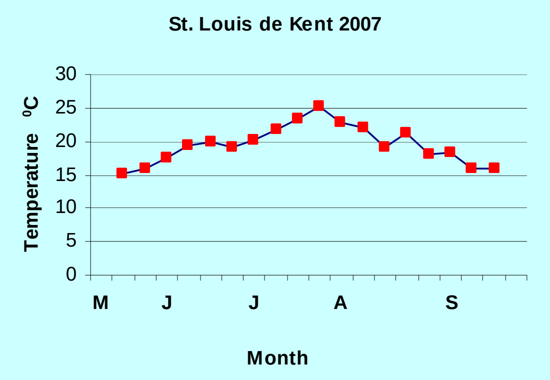
Each estuary has its own temperature characteristics. **Table 6** reflects the average temperature on the day of sampling after averaging all six stations in the estuary. There are individual differences depending on whether the sample site is inner, middle or outer estuary as is reflected in the standard deviation values. Temperature was obviously related to seasonal increases and decreases in air temperatures as spring moved to fall; hence the warmest temperatures occurred in July and August. Also, warmest air temperature in the early afternoon could affect water temperature later in the afternoon. Examining the data from 2006 and 2007, a couple of patterns are evident. In 2007, generally the May and June average temperature was slightly less than in 2006 but became slightly higher in July and August and remained about the same in September. In most cases the temperature difference was small, often less than one degree centigrade. Going back to the first year of CAMP data (Weldon et al 2005), the few locations that have continuous temperatures recorded do show an obvious increase in almost every location for the months compared.

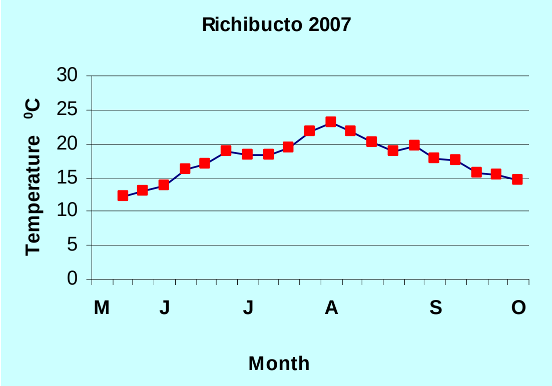
In 2007, one Vemco continuous temperature minilog probe was deployed in each CAMP estuary/coastal location. In most cases, the loggers were attached to a wooden stake and then screwed into a floating dock structure. Recovery rates were slightly less than 2006, and there was a new learning experience. The bivalve *Teredo navalis* (common name shipworm) was a very effective infiltrator of the wooden stakes. In the short CAMP season they were very efficient in penetrating the untreated wooden stake and their network of burrowed holes weakened the structure just below the water level. Subsequent strong wind and waves would cause the stakes to break off in many locations. Many stakes were recovered and would have been lost in another month due to the shipworm's activity. In 2008 this will be addressed by using metal stakes.

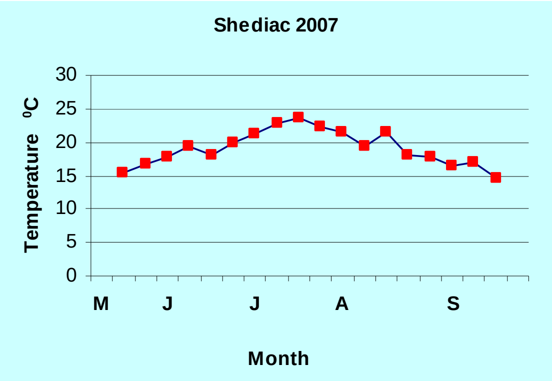
The loggers were set to record at hourly intervals and the graphed data represent weekly averages. The following graphs represent those weekly averages for those locations where probe information was available.

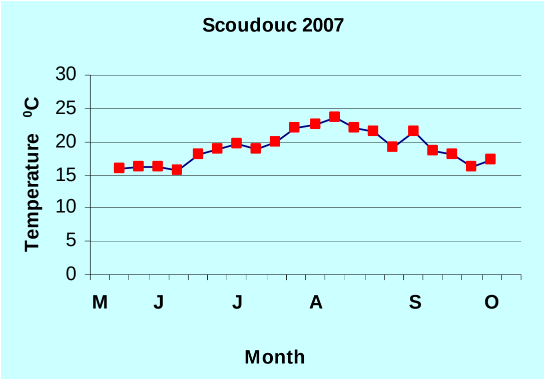


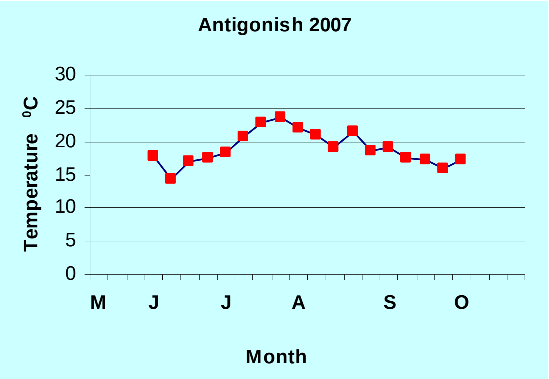


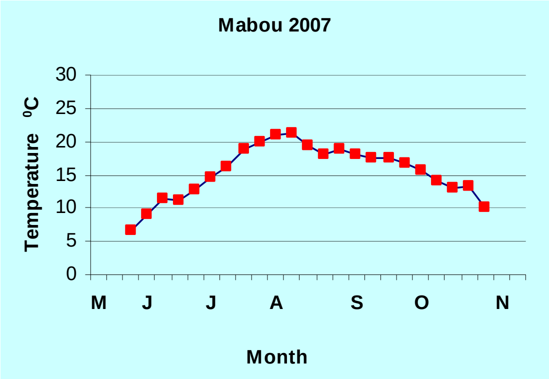


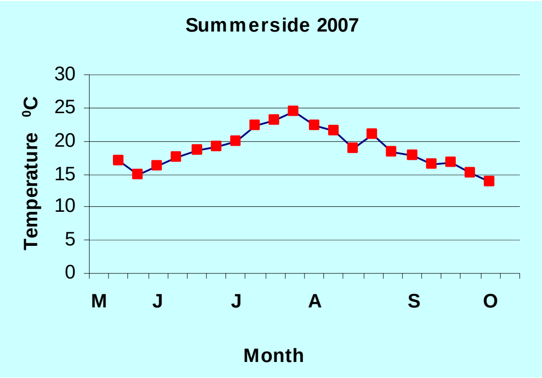


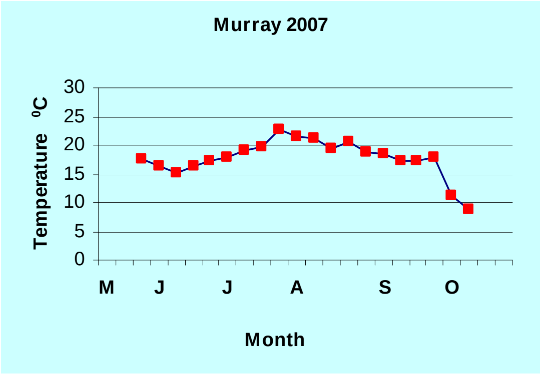












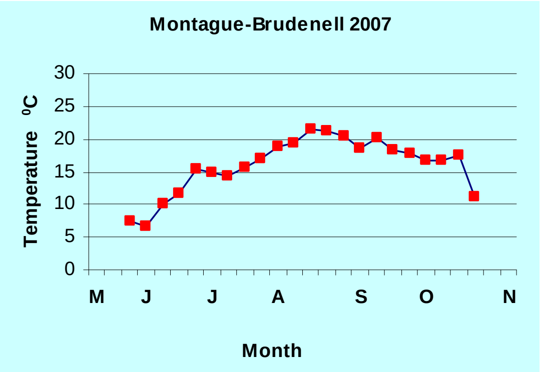


Figure 8. Graphs representing the weekly mean temperature (^{o}C) determined from hourly readings from Vemco minilog temperature recorders for all sites involved in the 2007 sampling season for CAMP

In New Brunswick, three of the six locations show slight overall increases in surface temperature from the comparable locations in 2006. Shediac was slightly lower than 2006 and Richibucto remained about the same. All six locations show a drop of about two degrees centigrade in mid-August followed by an increase to previous levels in late August-early September. If this decline were correlated to weather at that time, it would seem there was a cold spell in that two week period. It is not practical to bring weather variables into this analysis at this time. Noticeable is the slight differences between the adjacent Scoudouc and Shediac rivers possibly attributable to one location being farther out at the mouth of the estuary (Shediac) compared to the inner location of the other (Scoudouc). All six graphs showed a gradual spring increase in surface water temperature to a high in early August followed by the expected decline as air temperatures cool in September.

In Nova Scotia, only Mabou could be compared to 2006 values and no average temperature differences are apparent. The two locations, Antigonish being the other, did also show the same approximate two degree drop in mid-August as the sites in New Brunswick. The probe was also in the water longer in Mabou.

In Prince Edward Island, minilog recovery was poor also with only two from the east and the one in Summerside being returned. The only comparable location, Montague-Brudenell was slightly warmer in 2007 than 2006. The same drop in temperature as the other two provinces occurred in mid August in Summerside and Murray locations but was two weeks later in Montague-Brudenell.

Values for salinity (**Table 7**) can vary extensively among sites at each location which can be noticed by examining the standard variation values. When monthly averages were examined monthly variations were not extreme. If the tide was just starting to recede in the upper estuary, the salinity would naturally be higher than if the tide had not come in yet due in part to greater influence from headwaters leaving the estuary. As much as possible, groups attempted to sample on a similar tide regime each month. Each year, their efforts in coordinating sampling times have improved. Stations are usually visited on a rising tide that continues to reach its fullest and starts to drop during the four to five hours it generally takes to complete all six CAMP stations. Since the groups try to start out sampling each morning, this regime sometimes has to be adjusted. There have been instances where sampling is carried out over two days. There are other circumstances, such as new and full moon tides that make the station unsatisfactory for a visit as the tide is too high. Drastic weather changes during the day can also make sampling uncomfortable or even possibly unsafe. One difference to note from the 2006 table is that Shediac River has been added in 2007. In past reports this location called Shediac was more precisely the Scoudouc River which it will be referred to for all future reference.

In the upper estuary samples, salinity was generally lower as expected due to the large fresh water influence. There is monthly variation. The dominant pattern is that early months have lower salinities for most locations, probably due to run-off related to spring snow melt. This was more noticeable in systems like Bouctouche, St. Louis, Tabusintac and the Miramichi rivers. Other locations were sampled more towards the mouth where this influence was less pronounced. Middle sites in most locations vary in salinity, and this probably can be attributed to the state of the tide. Incoming tides bring in higher salt water content as compared to the increase in fresh water outflow when the tide was receding. The outer stations had the higher salinities. The comparison of salinities from the two years was in close agreement for the last two years for all locations throughout the Gulf Region.

Table 7. Average monthly salinity (ppt ± S.D.) per location for the 2007 season (n = 6). (NA = not available)

	Salinity (ppt) ± S.D.				
N.B.	**May**	**June**	**July**	**Aug**	**Sept**
Caraquet	**24.3 ± 2.0**	**25.3 ± 1.5**	**25.7 ± 0.7**	**25.6 ± 1.6**	**26.5 ± 1.6**
Lamèque	**26.0 ± 1.1**	**26.3 ± 1.0**	**26.3 ± 2.0**	**26.0 ± 3.0**	**27.7 ± 0.4**
Shippagan	**27.6 ± 1.3**	**27.2 ± 0.6**	**26.5 ± 1.6**	**27.0 ± 2.0**	**28.5 ± 0.3**
Tracadie	**20.9 ± 3.6**	**24.3 ± 2.2**	**26.1 ± 1.8**	**25.3 ± 2.8**	**27.4 ± 0.3**
Tabusintac	**13.5 ± 2.8**	**18.2 ± 4.5**	**19.7 ± 4.4**	**22.7 ± 2.9**	**23.5 ± 1.9**
Miramichi	**13.9 ± 6.0**	**16.3 ± 5.8**	**18.8 ± 6.0**	**17.6 ± 6.4**	**18.5 ± 5.4**
St Louis	**7.7 ± 5.8**	**15.1 ± 4.3**	**15.1 ± 2.8**	**21.5 ± 2.2**	**22.1 ± 1.9**
Richibucto	**15.0 ± 3.2**	**25.4 ± 1.3**	**21.9 ± 2.1**	**25.9 ± 1.1**	**26.8 ± 1.2**
Bouctouche	**9.5 ± 2.4**	**24.0 ± 0.7**	**24.3 ± 0.6**	**23.9 ± 1.0**	**24.3 ± 0.6**
Cocagne	**16.0 ± 3.3**	**24.4 ± 3.0**	**23.0 ± 3.0**	**27.7 ± 0.6**	**27.5 ± 0.8**
Shediac	**18.8 ± 2.5**	**21.4 ± 2.3**	**25.2 ± 1.3**	**24.2 ± 2.3**	**21.8 ± 2.7**
Scoudouc	**22.0 ± 2.2**	**22.5 ± 1.3**	**25.2 ± 0.6**	**27.0 ± 0.5**	**27.7 ± 0.2**
Jourimain	**27.7 ± 0.5**	**26.8 ± 1.1**	**27.8 ± 0.2**	**28.2 ± 0.1**	**28.2 ± 0.2**
Average NB	**18.7 ± 2.8**	**22.9 ± 2.3**	**23.5 ± 2.1**	**24.8 ± 2.0**	**25.4 ± 1.3**
N.S.					
Philip	**N/A**	**25.0 ± 2.3**	**28.0 ± 0.6**	**25.0 ± 2.4**	**28.2 ± 0.8**
Pugwash	**21.9 ± 5.4**	**22.6 ± 5.6**	**27.8 ± 0.8**	**29.0 ± 0.3**	**26.6 ± 0.7**
Pictou	**19.5 ± 3.7**	**13.0 ± 8.5**	**24.2 ± 6.3**	**23.3 ± 2.8**	**28.1 ± 0.8**
Antigonish	**20.3 ± 5.0**	**14.4 ± 7.7**	**19.9 ± 5.1**	**18.4 ± 5.0**	**26.5 ± 2.5**
Mabou	**20.8 ± 1.1**	**16.6 ± 9.1**	**19.4 ± 7.9**	**2.7 ± 0.8**	**16.0 ± 2.9**
Average NS	**20.6 ± 3.8**	**18.3 ± 6.6**	**23.9 ± 4.1**	**19.7 ±2.3**	**25.1 ± 1.5**
P.E.I.					
Mill River	**18.9 ± 2.0**	**23.2 ± 1.8**	**24.8 ± 0.9**	**25.1 ± 1.2**	**25.1 ± 0.9**
Trout River	**15.9 ± 9.5**	**18.9 ± 7.5**	**24.1 ± 2.4**	**22.0 ± 5.6**	**24.7 ± 4.3**
Summerside	**24.1 ± 5.2**	**21.5 ± 7.9**	**19.5 ± 8.7**	**23.7 ± 3.8**	**24.7 ± 2.9**
Basin Head	**28.9 ± 0.5**	**26.1 ± 2.3**	**28.4 ± 0.3**	**23.2 ± 2.0**	**28.4 ± 0.8**
Mont.-Bruden	**28.1 ± 0.2**	**26.9 ± 1.4**	**27.6 ± 0.8**	**27.1 ± 1.0**	**28.3 ± 0.3**
Murray	**27.5 ± 0.9**	**27.2 ± 1.3**	**28.3 ± 1.1**	**25.4 ± 8.5**	**27.5 ± 0.9**
Pinette	**26.9 ± 0.8**	**27.9 ± 0.4**	**28.0 ± 0.4**	**26.9 ± 0.6**	**26.1 ± 1.3**
Average PEI	**24.3 ± 2.7**	**24.5 ± 3.2**	**25.8 ± 2.1**	**24.8 ± 3.2**	**26.4 ± 1.6**

In New Brunswick, lower salinities were noted for St. Louis de Kent and Miramichi estuaries because they were sampled further up the estuary than other sites in NB. These two longer estuaries had lower salinities in the upper sites and higher salinities at the mouth. The stations located farther up river were changed in 2007 and moved downriver where the salinity range is comparable to the other locations.

Dissolved oxygen values (**Table 8**) were taken at each station after completion of the beach sampling. The average of the six stations was used to provide a value for that sampling location for that month and summarized in the table below.

Table 8. Average monthly dissolved oxygen (mg/l ± S.D.) per location for the 2007 season (n = 6). (NA = not available)

	Dissolved Oxygen (mg/l) ±S.D.				
N.B.	**May**	**June**	**July**	**Aug**	**Sept**
Caraquet	**10.5 ± 1.8**	**9.1 ± 1.4**	**8.9 ± 1.4**	**9.2 ± 1.8**	**8.9 ± 0.2**
Lamèque	**9.1 ± 0.4**	**10.6 ± 4.2**	**5.0 ± 0.6**	**7.3 ± 1.6**	**6.5 ± 1.4**
Shippagan	**9.3 ± 0.3**	**8.9 ± 1.0**	**10.2 ± 1.0**	**12.8 ± 3.4**	**9.0 ± 1.7**
Tracadie	**9.1 ± 0.6**	**8.0 ± 1.2**	**9.0 ± 2.0**	**9.8 ± 1.2**	**6.9 ± 1.9**
Tabusintac	**9.1 ± 1.1**	**6.9 ± 0.7**	**7.8 ± 0.7**	**7.3 ± 1.1**	**6.4 ± 0.8**
Miramichi	**13.0 ± 0.6**	**10.6 ± 0.5**	**10.8 ± 1.2**	**11.6 ± 1.2**	**12.1 ± 1.0**
St Louis	**8.2 ± 3.3**	**8.5 ± 0.5**	**8.1 ± 1.7**	**7.6 ± 0.4**	**8.9 ± 0.8**
Richibucto	**10.7 ± 0.3**	**8.3 ± 0.6**	**9.3 ± 1.5**	**8.4 ± 1.4**	**9.4 ± 0.9**
Bouctouche	**10.1 ± 0.6**	**6.8 ± 0.4**	**6.6 ± 0.7**	**9.2 ± 6.7**	**6.6 ± 0.7**
Cocagne	**10.3 ± 0.2**	**8.9 ± 1.0**	**9.8 ± 1.6**	**6.8 ± 0.5**	**6.8 ± 0.9**
Shediac	**9.5 ± 2.1**	**8.8 ± 1.9**	**5.9 ± 1.4**	**8.7 ± 1.4**	**10.6 ± 1.8**
Scoudouc	**8.7 ± 0.3**	**10.3 ± 2.0**	**8.0 ± 1.8**	**6.4 ± 0.8**	**5.9 ± 0.5**
Jourimain	**8.3 ± 1.5**	**6.6 ± 0.9**	**7.1 ± 0.4**	**8.2 ± 1.5**	**7.9 ± 0.4**
Average NB	**9.7 ± 1.0**	**8.6 ± 1.3**	**8.2 ± 1.2**	**8.7 ± 1.8**	**8.1 ± 1.0**
N.S.					
Philip	**N/A**	**8.9 ± 0.4**	**9.0 ± 1.0**	**9.7 ± 1.9**	**7.3 ± 0.5**
Pugwash	**10.0 ± 0.4**	**9.5 ± 0.6**	**8.9 ± 0.7**	**9.2 ± 0.4**	**10.2 ± 2.0**
Pictou	**10.0 ± 1.6**	**9.6 ± 2.6**	**9.7 ± 1.9**	**9.0 ± 0.7**	**11.1 ± 0.6**
Antigonish	**10.7 ± 0.7**	**11.0 ± 2.0**	**9.0 ± 1.4**	**9.1 ± 1.5**	**13.4 ± 3.1**
Mabou	**10.4 ± 0.2**	**9.0 ± 0.8**	**7.4 ± 0.4**	**7.9 ± 0.6**	**7.5 ± 1.4**
Average NS	**10.3 ± 0.7**	**9.6 ± 1.3**	**8.8 ± 1.1**	**9.0 ± 1.0**	**9.9 ± 1.5**
P.E.I.					
Mill River	**10.6 ± 1.4**	**9.3 ± 1.3**	**6.2 ± 1.6**	**8.2 ± 2.4**	**6.5 ± 0.6**
Trout River	**13.0 ± 1.4**	**12.8 ± 2.6**	**8.5 ± 3.2**	**9.9 ± 4.2**	**7.8 ± 2.0**
Summerside	**8.6 ± 0.7**	**7.4 ± 0.8**	**8.1 ± 1.9**	**6.0 ± 0.6**	**9.2 ± 0.6**
Basin Head	**10.3 ± 1.6**	**9.6 ± 0.8**	**7.1 ± 1.0**	**5.8 ± 1.4**	**8.8 ± 1.7**
Mont.-Brudenell	**10.1 ± 0.7**	**8.0 ± 0.6**	**7.9 ± 0.4**	**9.1 ± 1.5**	**7.0 ± 0.3**
Murray	**9.4 ± 0.7**	**8.0 ± 0.6**	**7.3 ± 0.5**	**8.7 ± 0.7**	**6.1 ± 0.5**
Pinette	**8.7 ± 0.4**	**8.1 ± 0.3**	**6.8 ± 0.6**	**8.6 ± 0.3**	**9.5 ± 1.8**
Average PEI	**10.1 ± 1.0**	**9.0 ± 1.0**	**7.4 ± 1.3**	**8.0 ± 1.6**	**7.8 ± 1.1**

Dissolved oxygen values (**Table 8**) were generally highest in the spring sampling and had decreased by the June sampling then show a slight, though noticeable increase in August in most locations throughout the Gulf Region. Dissolved oxygen preferences for permanent estuary/coastal shoreline species encountered with CAMP sampling methodology is not well documented. The general pattern for oxygen levels in the estuary was higher quantities in the spring that fluctuated in the summer and generally increased again in the fall. Processes that allow oxygen input in the water column include photosynthesis by plants and mixing from the air due to turbulence. In locations where there was more organic matter, decomposition of organic matter could produce the lower oxygen levels recorded (Tchoukanova et al. 2003). Overall, most DO values are higher when compared to similar months and locations in 2006.

In New Brunswick, May values were down in the north and higher in the south than they were in 2006 (Weldon et al. 2007). For all five months, there were seven recordings above 10 mg/l in 2006 (four of those in May). In 2007, there were 14 recordings above 10 mg/l when comparing all locations in NB for the five months. Miramichi, Antigonish and Trout rivers had the highest oxygen values while Tabusintac, Mabou and Mill rivers had the lowest when comparing the full sampling season to the other locations in each province.

3.6 Nutrient Analysis

In 2007, each group was given 60 number coded 30 ml bottles to collect a water sample and a replicate at each station in their estuary or coastal location for each of the five months. The protocol for collection is outlined in Appendix 2 in Weldon et al. (2008). The groups were given a small cooler bag with an ice pack to keep samples cool until they returned to a location that had a freezer. These frozen samples were returned to DFO where the season's sample was sent to the Bedford Institute of Oceanography for nutrient analysis. For each sample the total micromoles of silicate, phosphate, nitrate (NO_2 + NO_3), ammonia (NH_3) and nitrite (NO_2) were determined. From the raw data the average was calculated for each baseline location.

A detailed breakdown of the preliminary trends of the five nutrient compounds collected in September 2006 is provided by Theriault and Courtenay 2008 (unpublished report). This report will only compare the results from September 2007 with September 2006 (Weldon et al 2008). The other four months will be summarized in Appendix 2. There will be gaps in the data that was the result of samples left at the BIO lab for analysis being un-noticed in the cooler until next day. However it turns out that only one gap for September exists so some preliminary general comparisons are possible.

In New Brunswick, out of 11 locations where results are available for the two years, increases and decreases are evident. A numerical difference of around 50% will be the arbitrary value for indicating a difference that denotes an increase or decrease. For silicates, increases in Lamèque, Shippagan and Scoudouc, decreases in Tabusintac, Miramichi, St. Louis and Richibucto occurred while Tracadie, Bouctouche, Cocagne and Jourimain remained about the same. Phosphates increased in Richibucto, decreased in Scoudouc and remained about the same in all other locations. Except for increases in Lamèque, Shippagan, Miramichi and Richibucto, nitrates remained about the same in all other locations. A noteworthy drop in ammonia in Scoudouc, smaller drops in Miramichi and Cocagne, and an increase in Richibucto are apparent while the other locations remain about the same. A nitrite increase in Richibucto, decreases in St. Louis and Scoudouc compare to the other locations remaining at about he same levels as 2006.

In Nova Scotia, overall, there were decreases for all nutrients in all locations except in Antigonish where increases were recorded for all nutrients except nitrite that remained about the same. The other exception was a slight increase for ammonia in Pugwash and River Philip. The level of the decreases (excluding Antigonish) exceeded the 50% level in 14 of 18 possible nutrient-location combinations. This group (>50%) includes all the decreases in silicates in Philip, Pugwash, Mabou and Pictou, in phosphates in Mabou and Pictou, in nitrates in Mabou, in ammonia in Philip, Pugwash, Mabou and Pictou and nitrites in Philip, Pugwash and Mabou. Decreases of less than 50% in ammonia in Philip, Pugwash and Pictou, and of nitrite in Pictou were the other values noted in **Table 9**.

In Prince Edward Island, overall levels of all nutrients were down everywhere compared to 2006. Any increases noted were for phosphates and nitrates only, and most were slight. These include a phosphate increase of less than 50% in Trout River, Basin Head and Montague-Brudenell and a similar less than 50% increase in nitrates in Basin Head and Pinette locations. Nitrate levels had considerably decreased in Mill River, 101.33 µM/L (2006) to 1.39 µM/L (2007) and this was the biggest drop noted for any of the CAMP data in the whole Gulf Region.

Table 9. Average nutrient content of five listed compounds (µM/L ± S.D.) per location for the 2007 season (n = 12). (µM/L = µg atom /L) (NA = not available)

	Silicate	Phosphate	Nitrate	Ammonia	Nitrite
	µM/L ± SD	µM/L ± SD	µM/L ± SD	µM/L ± SD	µM/L ± SD
N.B.					
Caraquet 2006	4.12 ± 0.57	0.37 ± 0.04	0.71 ± 0.04	0.87 ± 0.11	0.17 ± 0.00
2007	NA	NA	NA	NA	NA
Lamèque	3.37 ± 0.85	0.41 ± 0.10	0.41 ± 0.15	4.45 ± 1.58	0.19 ± 0.04
	6.69 ± 3.50	0.81 ± 0.45	1.65 ± 0.59	6.07 ± 6.47	0.14 ± 0.13
Shippagan	1.08 ± 0.32	0.35 ± 0.07	0.39 ± 0.12	1.40 ± 0.37	0.12 ± 0.03
	3.33 ± 5.22	0.63 ± 0.41	1.33 ± 1.62	2.13 ± 1.30	0.16 ± 0.08
Tracadie	4.29 ± 0.96	0.55 ± 0.06	0.13 ± 0.04	1.60 ± 0.31	0.05 ± 0.01
	6.83 ± 2.32	0.73 ± 0.40	1.12 ± 0.21	1.58 ± 0.52	0.13 ± 0.03
Tabusintac	16.07 ± 3.04	0.67 ± 0.04	0.88 ± 0.13	1.70 ± 0.61	0.15 ± 0.01
	9.57 ± 2.36	0.59 ± 0.17	1.30 ± 0.88	1.85 ± 0.74	0.17 ± 0.11
Miramichi	25.11 ± 6.70	0.56 ± 0.04	0.54 ± 0.14	3.31 ± 0.46	0.17 ± 0.03
	13.05 ± 10.36	0.94 ± 0.33	1.19 ± 0.28	1.39 ± 0.63	0.14 ± 0.05
St Louis	13.54 ± 2.35	0.74 ± 0.09	1.31 ± 0.48	1.73 ± 0.35	0.19 ± 0.02
	5.57 ± 2.03	0.85 ± 0.13	1.05 ± 0.20	1.50 ± 0.81	0.04 ± 0.06
Richibucto	6.28 ± 1.83	0.36 ± 0.05	0.47 ± 0.39	1.13 ± 0.14	0.09 ± 0.01
	3.72 ± 3.59	2.00 ± 2.31	6.85 ± 7.33	8.75 ± 12.55	0.88 ± 1.14
Bouctouche	5.53 ± 0.53	1.05 ± 0.05	0.86 ± 0.14	1.78 ± 0.17	0.18 ± 0.01
	7.61 ± 3.82	0.92 ± 0.47	1.17 ± 3.07	2.13 ± 1.47	0.14 ± 0.371
Cocagne	1.82 ± 0.35	0.53 ± 0.01	0.91 ± 0.10	5.80 ± 2.61	0.21 ± 0.04
	1.99 ± 1.56	0.56 ± 0.35	0.83 ± 0.23	1.93 ± 2.56	0.13 ± 0.07
Shediac	new				
	7.70 ± 4.79	0.56 ± 0.10	2.42 ± 3.63	0.962 ± 0.41	0.11 ± 0.04
Scoudouc	3.05 ± 1.80	2.18 ± 1.32	1.08 ± 0.39	34.32 ± 30.09	0.21 ± 0.05
	1.67 ± 0.83	0.44 ± 0.20	0.80 ± 0.10	2.53 ± 1.42	0.11 ± 0.03
Jourimain	0.54 ± 0.04	0.45 ± 0.05	0.71 ± 0.03	0.60 ± 0.14	0.16 ± 0.01
	0.85 ± 0.22	0.49 ± 0.11	0.79 ± 0.18	0.79 ± 0.67	0.11 ± 0.04
N.S.					
R. Philip 2006	6.08 ± 1.74	0.62 ± 0.13	0.91 ± 0.31	2.69 ± 0.21	0.23 ± 0.04
2007	2.51 ± 1.36	0.69 ± 0.05	0.65 ± 0.03	0.92 ± 0.68	0.08 ± 0.04
Pugwash	6.08 ± 1.74	0.62 ± 0.13	0.91 ± 0.31	2.69 ± 0.21	0.23 ± 0.04
	1.35 ± 1.38	0.68 ± 0.12	0.73 ± 0.08	1.16 ± 0.91	0.09 ± 0.03
Antigonish	2.51 ± 0.46	0.51 ± 0.12	0.09 ± 0.07	2.84 ± 0.55	0.12 ± 0.02
	4.59 ± 2.47	0.60 ± 0.10	0.58 ± 0.50	3.73 ± 2.88	0.11 ± 0.01
Mabou	22.66 ± 3.24	0.18 ± 0.02	2.30 ± 0.48	2.83 ± 0.44	0.20 ± 0.01
	10.69 ± 4.20	0.004 ± 0.01	0.69 ± 0.10	0.90 ± 0.32	0.06 ± 0.02
Pictou	7.30 ± 2.20	1.96 ± 1.09	1.89 ± 1.14	35.22 ± 32.71	0.52 ± 0.29
	3.80 ± 2.75	0.85 ± 0.20	1.09 ± 0.65	4.24 ± 6.74	0.20 ± 0.23

P.E.I.					
Mill River 2006	24.16 ± 1.93	0.38 ± 0.03	101.33 ± 12.63	10.50 ± 1.79	0.57 ± 0.07
2007	1.78 ± 1.46	0.37 ± 0.12	1.39 ± 0.59	2.26 ± 1.51	0.00 ± 0.00
Trout River	8.97 ± 2.74	0.54 ± 0.10	10.84 ± 3.64	4.02 ± 1.93	0.30 ± 0.07
	7.33 ± 7.56	0.79 ± 0.36	1.74 ± 1.92	1.69 ± 1.10	0.02 ± 0.04
Summerside	new				
	2.66 ± 1.13	0.52 ± 0.09	0.99 ± 0.07	5.40 ± 6.02	0.00 ± 0.00
Basin Head	3.02 ± 0.31	0.26 ± 0.02	1.86 ± 0.66	2.21 ± 0.29	0.12 ± 0.01
	2.80 ± 0.72	0.38 ± 0.16	2.47 ± 1.34	1.83 ± 0.54	0.02 ± 0.04
Pinette	12.53 ± 3.56	1.54 ± 0.23	0.72 ± 0.11	3.19 ± 0.44	0.20 ± 0.02
	8.58 ± 3.24	0.78 ± 0.11	1.38 ± 0.21	1.28 ± 0.78	0.12 ± 0.04
Murray	6.87 ± 1.71	0.83 ± 0.05	1.18 ± 0.84	1.21 ± 0.18	0.15 ± 0.02
	2.30 ± 1.86	0.49 ± 0.06	1.08 ± 0.33	1.00 ± 0.55	0.08 ± 0.05
Mont.-Bruden	1.02 ± 0.14	0.46 ± 0.04	1.00 ± 0.40	0.85 ± 0.05	0.22 ± 0.06
	0.45 ± 0.42	0.51 ± 0.07	1.02 ± 0.15	0.47 ± 0.20	0.09 ± 0.02

In **Appendix 2,** the results for the other four months will be summarized. This was the first year that water samples were taken for all five months for CAMP locations. The results are not totally complete as some samples were not analyzed. This will be indicated in the table. Some locations were lost completely and others were incomplete which would be reflected in averages being calculated with less than 12 samples at some locations.

Appendix 3 will once again provide summary pie charts of total percentages of the most abundant species at each location for the whole season (May to September). Because the maps showing samples sites are readily available in all previous reports, only the new locations will be included in this report. To find past reports, the following links to library archives will be useful:

2004	http://www.dfo-mpo.gc.ca/Library/319437.pdf
2005	http://www.dfo-mpo.gc.ca/Library/329182.pdf
2006	http://www.dfo-mpo.gc.ca/Library/332000.pdf

5.0 CONCLUSION

The fundamental objective of the CAMP program continues to be the development and continuation of an outreach program for DFO oceans and habitat staff to liaise with and engage the coastal communities in learning more about their estuaries and bays. The initial goal of developing a monitoring program that is NGO friendly has been realized. Baseline data have been collected for four years from a wide range of estuaries and bays in the Gulf of St. Lawrence. Full baseline CAMP locations sampled in 2007 increased from 18 to 24.

After four years of data collection, we have the minimum baseline information to test the hypothesis that animal assemblages can contribute to our knowledge about the health of these habitats. This analysis will be done after data from the fifth year (2008) is complete. The next steps for CAMP are now in the hands of the DFO and their associated university collaborators. The baseline data resulting from community efforts will be used to test the hypothesis that the numbers and types of near shore animals provide us with an indication of the overall health of these estuaries. The analysis of five years of data is planned to be available in the fall of 2009. Hypothesis such as whether the absence or presence or abundance of particular species reflect particular environmental problems or is it overall species diversity that provides the best single metric for estuarine health? When these and related questions have been addressed and the resulting conclusions have been returned to the participating community groups and the public at large, CAMP will have fulfilled its immediate objectives. However, it is recognized that as the CAMP evolves, there is always room for additional refinement. As such, the commitment to consultation with all stakeholders will continue.

NGO’s anticipate that the data they have gathered will be helpful in developing a useful tool to assist them to monitor the health and condition of their estuary. Improvements such as the enhanced participation of the Coalition ensure that gathered data is available to groups to assist them in planning future direction, identifying local areas of concern and determining the present status of their estuary under their mandate.

Besides a commitment to quality data recording, the community groups realize they have only had to contribute resources for one day a month to acquire these data. The integrated approach provided by the University partners, DFO and certain funding agencies has helped guide the development of the CAMP to a direction that will maximize output goals from minimal input.

There is an expectation that more specialized science could easily be developed from the outcomes of the present CAMP data. As various models of watershed management becomes more widely used, CAMP protocols and gathered information will become a key component in the overall management of a watershed.

ACKNOWLEDGEMENTS

This program was originally developed through the Oceans and Habitat Division, Oceans and Science Branch (OSB), Gulf Region DFO. Gulf Region NGO's remain onboard with tremendous enthusiasm for volunteers that oversee ecological programs across the region. Summer students, both employed and volunteering were an invaluable resource for data collection. Some funding for student salaries came from Interdisciplinary Studies in Aquatic Resources at St. Francis Xavier University, and grants to Dr. David Garbary from the Natural Sciences and Engineering Research Council of Canada. Other groups used portions of Environmental Trust funds to free up resources for the one day a month necessary to maintain the program.

A strengthened involvement by the Southern Gulf of St. Lawrence Coalition on Sustainability has provided a common link to all NGOs in the Gulf Region. They have and will continue to coordinate students to provide the NGOs with experienced manpower and vehicular assistance.

Also thanks to high school student Holly Vail for her help with sediment analysis during the winter as part of her Coop internship with DFO.

After the initial set-up of the pilot stage by several of the authors, the continued support of all stakeholders and continued consultation throughout the 2007 sampling season ensured that the program maintained strong links to the original concept.

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Appendix 1. List of Species Collected during the CAMP 2007

fish

alewife (gaspereau) (*Alosa sp.***)**
American sand lance (*Ammodytes hexapterus***)**
Atlantic silverside (*Menidia menidia***)**
Atlantic tomcod (*Microgadus tomcod***)**
banded killifish (*Fundulus diaphanus***)**
black spotted stickleback (*Gasterosteus wheatlandi***)**
blueback herring (*Alosa aestivalis***)**
brook (speckled) trout (*Salvelinus fontinalis***)**
cunner (*Tautogolabrus adspersus***)**
fourspine stickleback (*Apeltes quadracus***)**
ninespine stickleback (*Pungitius pungitius***)**
northern pipefish (*Syngnathus fuscus***)**
mummichog (*Fundulus heteroclitus***)**
rainbow smelt (*Osmerus mordax***)**
rock gunnel (*Pholis gunnellus***)**
shorthorn sculpin (*Myoxocephalus scorpius***)**
smooth flounder (*Pleuronectes putnami***)**
striped bass (*Morone saxatilis***)**
threespine stickleback (*Gasterosteus aculeatus***)**
white perch (*Morone americanus***)**
windowpane flounder (*Scophthalmus aquosus***)**
winter flounder (*Pseudopleuronectes americanus***)**
American Eel (*Anguilla rostrata***)**
Atlantic salmon parr (*Salmo salar***)**
Atlantic mackerel (*Scomber scombus***)**
Rainbow trout (*Salmo gairdneri***)**
White hake (*Urophycis tenuis***)**
Trout sp (*Salmo species***)**

Invertebrates

Clubbed tunicate (*Styela clava***)**
Violet tunicate (*Betrylloides violaceus***)**
Vase tunicate (*Ciona intestinalis***)**

crustaceans

grass shrimp
(*Palaemonetes vulgaris***)**
green crab
(*Carcinus maenas***)**
lady crab
(*Ovalipes ocellatus***)**
mud crabs
(*Xanthidae sp.***)**
rock crab
(*Cancer irroratus***)**
sand shrimp
(*Crangon septemspinosa***)**

Appendix 2 Average nutrient content of five listed compounds (µM/L ± S.D.) per location for the 2007 season (n = 12) for the months of May, June, July and August. (µM/L = µg atom /L) (n/a = not available).

* means not all samples available			Silicate	Phosphate	Nitrate	Ammonia	Nitrite
			µM/L±SD	µM/L±SD	µM/L±SD	µM/L±SD	µM/L±SD
N.B.							
Caraquet		May	n/a	n/a	n/a	n/a	n/a
		June	n/a	n/a	n/a	n/a	n/a
		July	n/a	n/a	n/a	n/a	n/a
		Aug	n/a	n/a	n/a	n/a	n/a
Lamèque		May	2.93 ± 1.75	0.30 ± 0.39	0.67 ± 0.29	3.61 ± 3.52	0.15 ± 0.08
		June	8.11 ± 4.45	1.55 ± 0.24	0.78 ± 0.48	3.12 ± 1.34	0.19 ± 0.01
		July	10.70 ± 8.39	1.64 ± 0.50	1.17 ± 0.91	5.44 ± 7.67	0.26 ± 0.12
		Aug	14.39 ± 14.97	1.22 ± 0.59	1.77 ± 2.48	6.14 ± 12.93	0.29 ± 0.13
Shippagan		May	2.04 ± 4.28	0.36 ± 0.23	1.25 ± 2.69	3.11 ± 1.75	0.17 ± 0.13
		June	2.79 ± 3.62	0.53 ± 0.20	0.68 ± 2.24	1.45 ± 1.67	0.11 ± 0.11
		July	14.13 ± 6.05	1.32 ± 0.47	1.38 ± 1.98	2.14 ± 1.51	0.14 ± 0.09
		Aug	4.53 ± 5.62	0.87 ± 0.44	0.84 ± 1.78	2.02 ± 1.40	0.15 ± 0.09
Tracadie		May	10.61 ± 5.96	0.49 ± 0.24	0.79 ± 0.19	1.97 ± 1.24	0.13 ± 0.03
		June	7.02 ± 3.17	0.53 ± 0.15	0.68 ± 0.02	0.84 ± 0.35	0.07 ± 0.01
		July	6.47 ± 2.50	1.07 ± 0.42	0.99 ± 0.03	1.10 ± 0.52	0.11 ± 0.02
		Aug	5.04 ± 1.61	1.08 ± 0.51	1.04 ± 0.05	1.17 ± 0.39	0.11 ± 0.03
Tabusintac		May	n/a	n/a	n/a	n/a	n/a
		June	15.52± 7.29	0.54 ± 0.44	0.86 ± 0.16	1.75 ± 0.96	0.12 ± 0.02
		July	10.14 ± 3.33	0.54 ± 0.44	0.71 ± 0.07	1.05 ± 0.61	0.09 ± 0.01
		Aug	7.49 ± 1.71	0.83 ± 0.38	0.90 ± 0.04	1.94 ± 0.64	0.11 ± 0.02
Miramichi		May	n/a	n/a	n/a	n/a	n/a
		June	18.13 ± 9.57	0.40 ± 0.15	1.09 ± 0.36	1.75 ± 1.17	0.17 ± 0.08
		July	n/a	n/a	n/a	n/a	n/a
		Aug	16.21 ± 10.35	0.44 ± 0.14	0.99 ± 0.13	1.15 ± 0.64	0.13 ± 0.04
St Louis	*	May	10.58 ± 6.74	0.26 ± 0.11	2.14 ± 0.98	2.68 ± 0.27	0.21 ± 0.05
	*	June	21.43 ± 2.35	0.22 ± 0.04	1.14 ± 0.24	1.16 ± 0.04	0.13 ± 0.00
		July	n/a	n/a	n/a	n/a	n/a
		Aug	7.86 ± 1.70	1.30 ± 0.27	0.84± 0.04	1.27 ± 0.52	0.16 ± 0.54
Richibucto		May	18.07 ± 8.95	0.05 ± 0.39	0.80 ± 0.30	1.43 ± 1.08	0.12 ± 0.07
	*	June	2.42 ± 0.48	0.31 ± 0.01	0.79 ± 0.01	2.95 ± 2.32	0.05 ± 0.00
		July	n/a	n/a	n/a	n/a	n/a
		Aug	3.57 ± 3.85	0.75 ± 0.31	1.00 ± 0.14	7.82 ± 9.31	0.15 ± 0.54

Appendix 2 (continued)

* means not all samples available		Silicate	Phosphate	Nitrate	Ammonia	Nitrite
		µM/L±SD	µM/L±SD	µM/L±SD	µM/L±SD	µM/L±SD
Bouctouche	May	14.25 ± 2.54	0.45 ± 0.11	5.52 ± 4.77	4.51 ± 2.14	0.30 ± 0.06
	June	8.12 ± 1.13	0.79 ± 0.29	1.05 ± 0.36	3.43 ± 2.60	0.22 ± 0.15
	* July	n/a	n/a	n/a	n/a	n/a
	Aug	7.50 ± 5.04	1.64 ± 0.17	2.83 ± 4.32	3.30 ± 1.50	0.19 ± 0.54
Cocagne	May	9.18 ± 2.72	0.28 ± 0.15	1.04 ± 0.12	2.12 ± 2.27	0.20 ± 0.09
	June	6.23 ± 3.98	0.45 ± 0.21	0.47 ± 0.33	0.60 ± 0.14	0.11 ± 0.00
	July	7.04 ± 0.16	0.57 ± 0.00	0.81 ± 0.00	0.60 ± 0.14	0.11 ± 0.00
	Aug	1.98 ± 1.65	1.06 ± 0.37	0.62 ± 0.23	2.26 ± 3.33	0.20 ± 0.09
Shediac	May	6.31 ± 4.30	0.21 ± 0.08	1.61 ± 1.92	1.71 ± 0.70	0.14 ± 0.04
	June	n/a	n/a	n/a	n/a	n/a
	* July	5.75 ± 0.97	1.61 ± 0.12	0.90 ± 0.08	2.53 ± 1.05	0.12 ± 0.02
	Aug	n/a	n/a	n/a	n/a	n/a
Scoudouc	May	4.01 ± 3.03	0.37 ± 1.92	0.72 ± 6.10	1.41 ± 10.60	0.19 ± 0.94
	June	5.06 ± 2.89	0.93 ± 1.67	0.92 ± 5.33	3.65 ± 9.36	0.16 ± 0.82
	* July	1.79 ± 0.14	0.53 ± 0.03	0.81 ± 0.02	1.32 ± 0.85	0.08 ± 0.02
	Aug	1.78 ± 0.79	3.41 ± 3.27	0.86 ± 0.23	4.07 ± 5.20	0.18 ± 0.47
Jourimain	May	n/a	n/a	n/a	n/a	n/a
	June	n/a	n/a	n/a	n/a	n/a
	July	n/a	n/a	n/a	n/a	n/a
	Aug	n/a	n/a	n/a	n/a	n/a
N.S.						
River Philip	May	n/a	n/a	n/a	n/a	n/a
	June	5.16 ± 2.24	0.21 ± 0.04	0.78 ± 0.05	2.17 ± 0.71	0.08 ± 0.02
	* July	2.84 ± 1.35	0.50 ± 0.20	0.79 ± 0.08	3.49 ± 2.41	0.08 ± 0.02
	Aug	4.69 ± 3.01	0.52 ± 0.14	0.76 ± 0.03	3.43 ± 1.19	0.08 ± 0.02
Pugwash	May	6.73 ± 3.92	0.22 ± 0.14	1.07 ± 0.35	2.70 ± 2.32	0.12 ± 0.03
	June	1.84 ± 0.80	0.36 ± 0.19	0.62 ± 0.04	1.52 ± 0.76	0.07 ± 0.01
	July	3.31 ± 1.46	0.70 ± 0.32	0.64 ± 0.07	1.69 ± 0.83	0.07 ± 0.03
	Aug	2.54 ± 1.48	0.70 ± 0.32	0.64 ± 0.07	1.69 ± 0.83	0.07 ± 0.03
Pictou	May	6.14 ± 2.30	1.96 ± 3.93	1.54 ± 0.92	21.14 ± 44.31	0.34 ± 0.44
	June	12.13 ± 5.98	0.79 ± 0.75	2.68 ± 2.51	7.42 ± 6.68	0.43 ± 0.52
	July	15.33 ± 8.52	3.78 ± 6.72	2.46 ± 1.94	6.24 ± 6.56	0.67 ± 0.96
	Aug	9.60 ± 5.93	1.51 ± 1.43	1.53 ± 1.17	11.87 ± 18.92	0.22 ± 0.18
Antigonish	May	5.72 ± 2.34	0.18 ± 0.13	0.30 ± 0.05	0.88 ± 0.72	0.06 ± 0.01
	June	15.96 ± 11.95	0.29 ± 0.17	0.75 ± 0.45	1.83 ± 0.90	0.14 ± 0.04
	July	6.41 ± 2.46	0.51 ± 0.31	0.45 ± 0.11	1.79 ± 1.41	0.08 ± 0.02
	Aug	16.31 ± 7.32	0.34 ± 0.14	0.71 ± 0.72	1.75 ± 0.73	0.12 ± 0.05
Mabou	May	7.43 ± 2.20	0.11 ± 0.04	0.93 ± 0.11	1.69 ± 0.37	0.07 ± 0.01
	June	10.93 ± 5.32	0.09 ± 0.07	0.95 ± 0.27	1.34 ± 0.44	0.11 ± 0.05
	July	7.69 ± 4.43	0.09 ± 0.05	0.68 ± 0.06	1.35 ± 0.54	0.15 ± 0.01
	Aug	8.73 ± 9.12	0.13 ± 0.07	1.05 ± 0.36	1.45 ± 0.41	0.10 ± 0.03

Appendix 2 (continued)

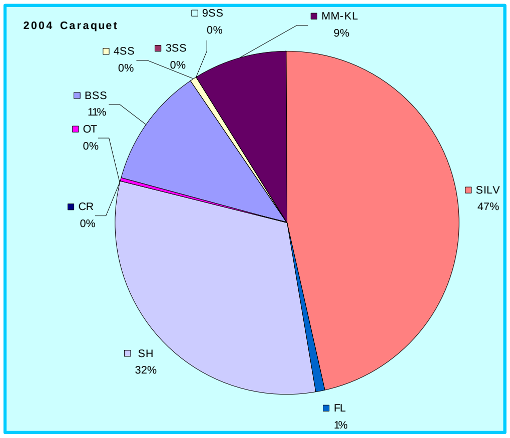
* means not all samples available		Silicate	Phosphate	Nitrate	Ammonia	Nitrite
		µM/L±SD	µM/L±SD	µM/L±SD	µM/L±SD	µM/L±SD
P.E.I.						
Mill River	May	12.57 ± 3.16	0.35 ± 0.25	68.10 ± 33.91	1.69 ± 0.73	0.90 ± 0.22
	June	12.82 ± 10.02	0.43 ± 0.19	29.44 ± 35.03	1.52 ± 1.15	0.33 ± 0.19
	July	5.69 ± 2.75	1.93 ± 0.81	2.82 ± 4.30	3.11 ± 4.81	0.29 ± 0.190
	Aug	12.42 ± 4.49	1.15 ± 0.24	2.82 ± 4.30	3.11 ± 4.81	0.29 ± 0.19
Trout River	May	11.94 ± 5.52	0.46 ± 0.18	24.44 ± 25.76	1.14 ± 0.94	0.39 ± 0.26
	June	14.98 ± 9.38	1.40 ± 0.92	10.69 ± 14.33	1.57 ± 1.08	0.05 ± 0.07
	July	8.05 ± 4.81	1.37 ± 0.57	0.71 ± 0.48	0.67 ± 0.20	0.12 ± 0.08
	Aug	11.16 ± 5.32	1.41 ± 0.51	3.66 ± 5.68	2.07 ± 1.38	0.17 ± 0.08
Summerside	May	4.30 ± 5.09	0.71 ± 0.67	38.73 ± 50.36	1.51 ± 0.92	0.57 ± 0.49
	June	4.67 ± 3.11	0.55 ± 0.29	8.94 ± 11.20	2.23 ± 1.39	0.36 ± 0.31
	July	5.48 ± 5.00	0.55 ± 0.38	31.81 ± 54.39	2.13 ± 2.38	0.70 ± 0.88
	Aug	7.42 ± 5.45	0.69 ± 0.20	22.41 ± 29.50	2.35 ± 1.85	0.70 ± 0.72
Basin Head	May	n/a	n/a	n/a	n/a	n/a
	June	5.03 ± 0.87	0.93 ± 0.53	6.02 ± 5.27	2.21 ± 0.91	0.25 ± 0.09
	July	2.76 ± 1.08	0.45 ± 0.23	1.71 ± 1.92	1.49 ± 0.77	0.12 ± 0.05
	Aug	9.94 ± 2.27	0.90 ± 0.66	9.92 ± 8.76	3.00 ± 2.16	0.28 ± 0.06
Pinette River	May	n/a	n/a	n/a	n/a	n/a
	June	2.49 ± 1.01	0.35 ± 0.10	0.82 ± 0.06	2.47 ± 1.22	0.00 ± 0.00
	July	3.91 ± 1.44	0.88 ± 0.24	0.48 ± 0.08	1.54 ± 1.07	0.09 ± 0.02
	Aug	9.75 ± 2.72	0.94 ± 0.15	0.85 ± 0.35	1.28 ± 0.78	0.12 ± 0.04
Murray River	May	3.23 ± 0.96	0.37 ± 0.08	0.83 ± 0.07	0.81 ± 0.55	0.09 ± 0.01
	June	4.99 ± 4.21	0.29 ± 0.11	1.04 ± 1.26	1.89 ± 2.95	0.06 ± 0.04
	July	5.96 ± 2.88	0.58 ± 0.09	0.56 ± 0.12	3.13 ± 2.81	0.08 ± 0.01
	Aug	2.80 ± 3.27	0.82 ± 0.61	0.46 ± 0.53	0.28 ± 1.21	0.06 ± 0.07
Mont.-Brud	May	3.54 ± 0.43	0.57 ± 0.17	0.43 ± 0.15	0.53 ± 0.35	0.07 ± 0.03
	June	0.86 ± 0.17	0.36 ± 0.09	0.69 ± 0.08	1.99 ± 1.24	0.01 ± 0.03
	July	2.59 ± 1.40	0.76 ± 0.18	1.26 ± 1.02	1.91 ± 2.32	0.13 ± 0.04
*	Aug	4.09 ± 2.31	0.90 ± 0.36	1.28 ± 1.59	2.12 ± 3.59	0.07 ± 0.02

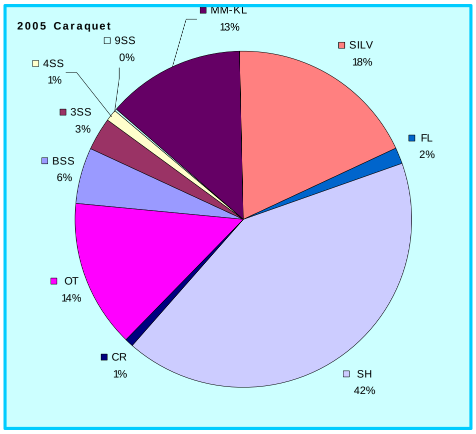
Appendix 3. Map of each Estuary/Coastal Shoreline Location Showing Sampling Sites plus four (if available) Pie Charts Summarizing the Season Total Percentages of the Most Abundant Species for the 2004, 2005, 2006 and 2007 Sampling Seasons.

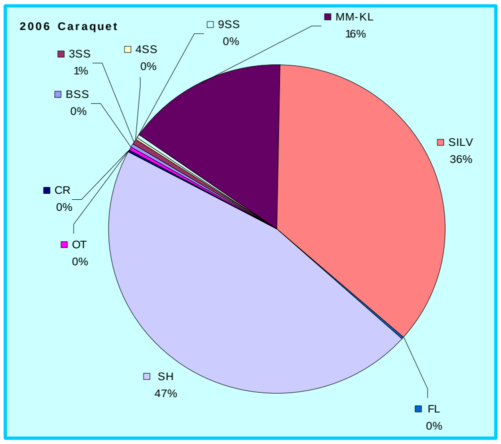
Caraquet / New Brunswick

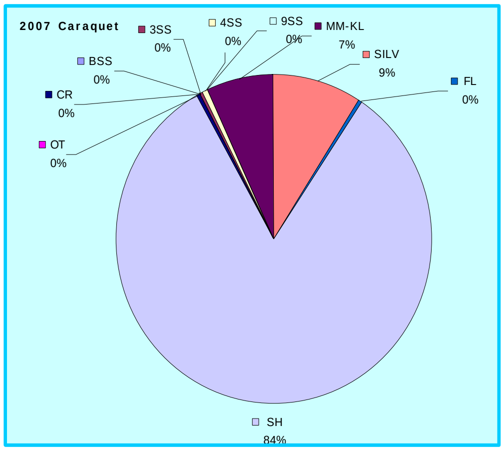
Black Spot Stickleback (BSS)
3 Spine Stickleback (3SS)
4 Spine Stickleback (4SS)
9 Spine Stickleback (9SS)
Mummichog/Killifish (MM-KL)

Atlantic Silverside (SILV)
Flounder Species (FL)
Shrimp Species (SH)
Crab Species (CR)
Other Species (OT)





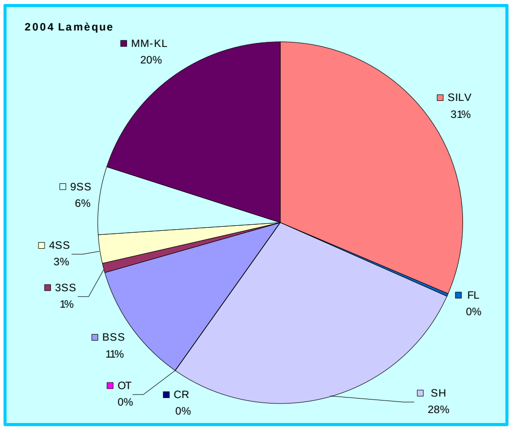


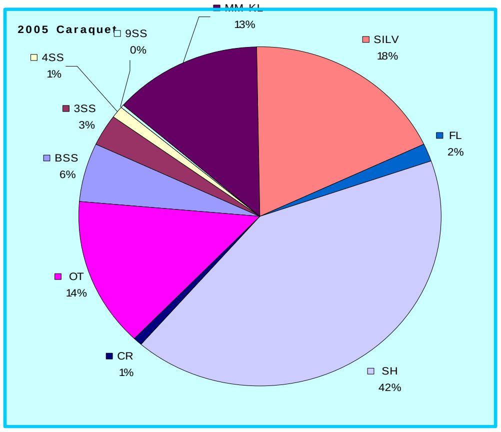


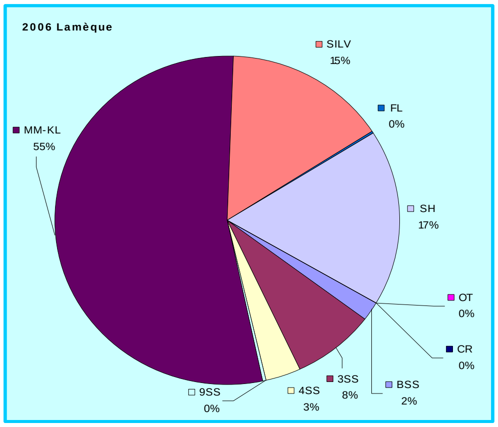
Lamèque / New Brunswick

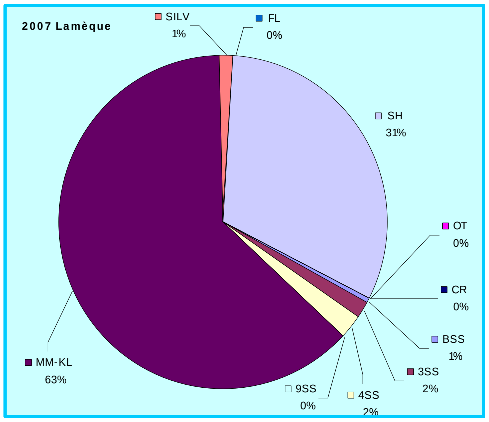
Black Spot Stickleback (BSS)
3 Spine Stickleback (3SS)
4 Spine Stickleback (4SS)
9 Spine Stickleback (9SS)
Mummichog/Killifish (MM-KL)

Atlantic Silverside (SILV)
Flounder Species (FL)
Shrimp Species (SH)
Crab Species (CR)
Other Species (OT)





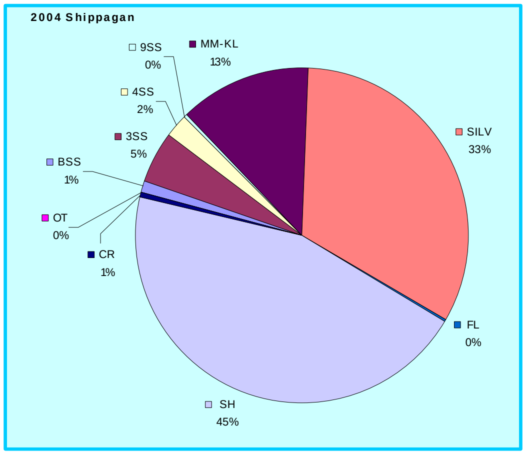


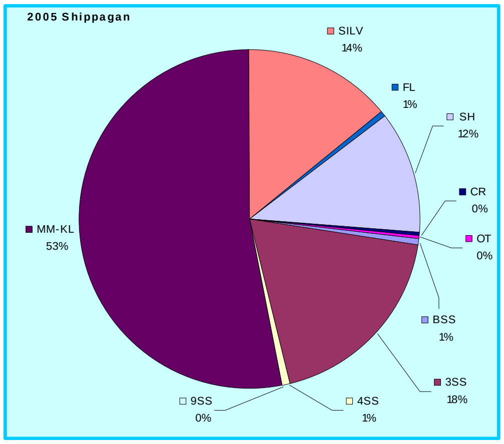


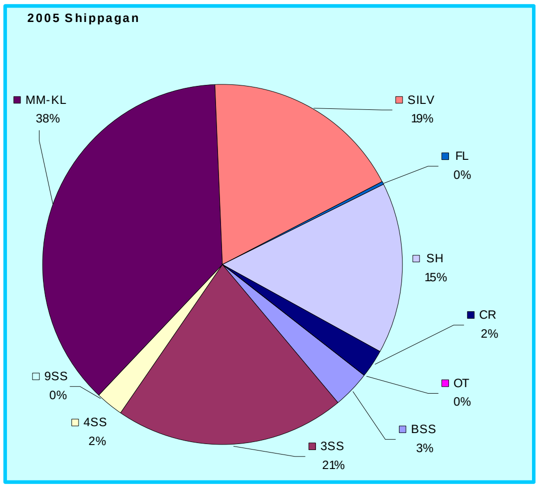
Shippagan / New Brunswick

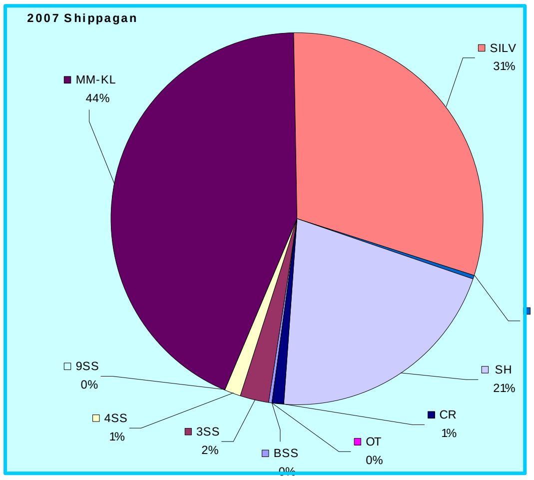
Black Spot Stickleback (BSS)
3 Spine Stickleback (3SS)
4 Spine Stickleback (4SS)
9 Spine Stickleback (9SS)
Mummichog/Killifish (MM-KL)

Atlantic Silverside (SILV)
Flounder Species (FL)
Shrimp Species (SH)
Crab Species (CR)
Other Species (OT)





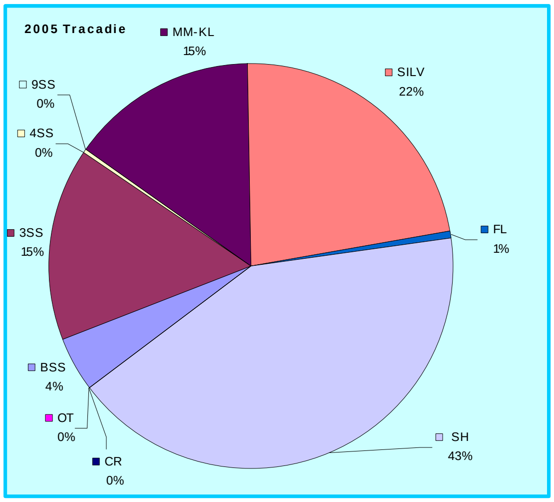


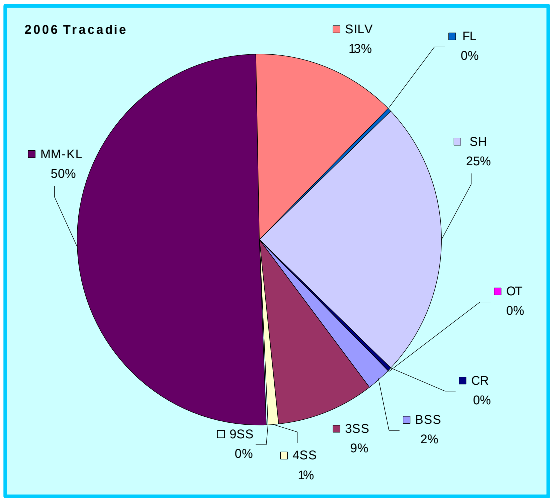


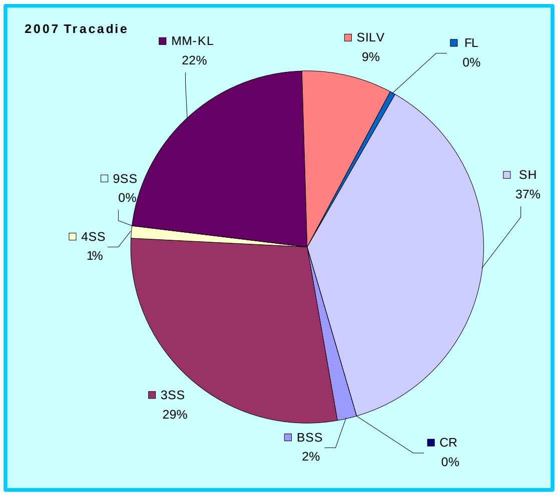
Tracadie / New Brunswick

Black Spot Stickleback (BSS)
3 Spine Stickleback (3SS)
4 Spine Stickleback (4SS)
9 Spine Stickleback (9SS)
Mummichog/Killifish (MM-KL)

Atlantic Silverside (SILV)
Flounder Species (FL)
Shrimp Species (SH)
Crab Species (CR)
Other Species (OT)

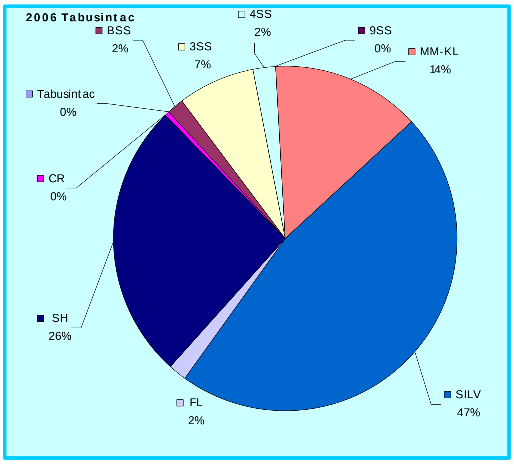


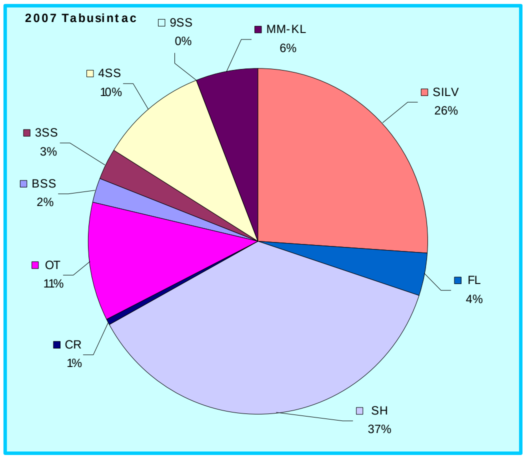




Tabusintac / New Brunswick

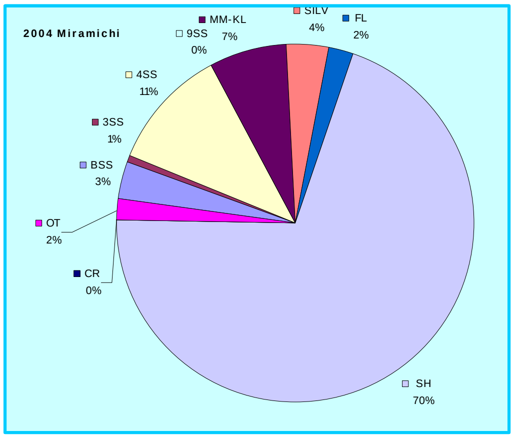
Black Spot Stickleback (BSS)
3 Spine Stickleback (3SS)
4 Spine Stickleback (4SS)
9 Spine Stickleback (9SS)
Mummichog/Killifish (MM-KL)
Atlantic Silverside (SILV)
Flounder Species (FL)
Shrimp Species (SH)
Crab Species (CR)
Other Species (OT)

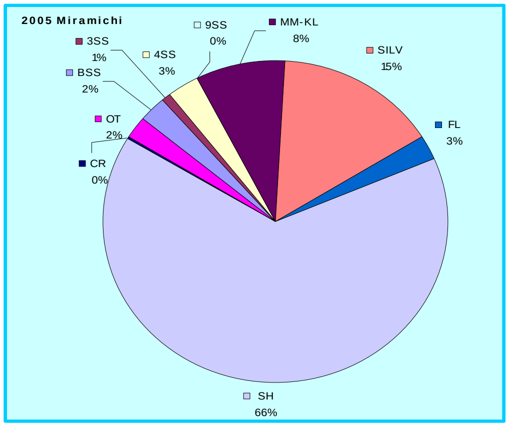


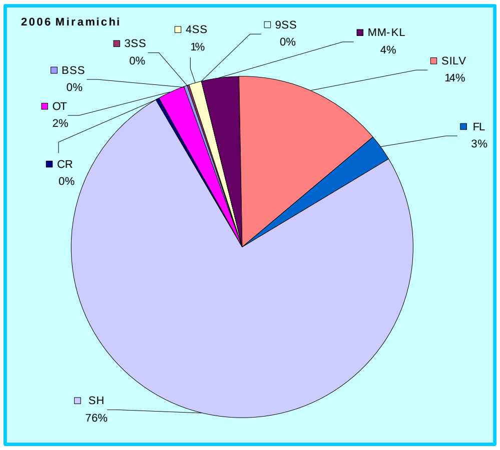


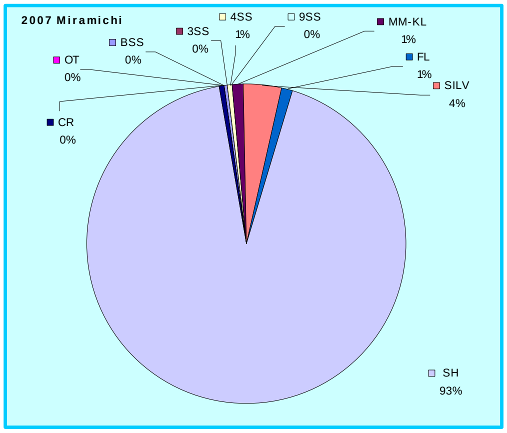
Black Spot Stickleback (BSS)
3 Spine Stickleback (3SS)
4 Spine Stickleback (4SS)
9 Spine Stickleback (9SS)
Mummichog/Killifish (MM-KL)
Atlantic Silverside (SILV)
Flounder Species (FL)
Shrimp Species (SH)
Crab Species (CR)
Other Species (OT)

Miramichi / New Brunswick





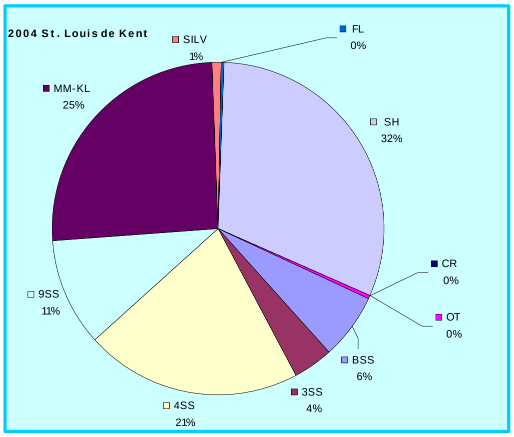


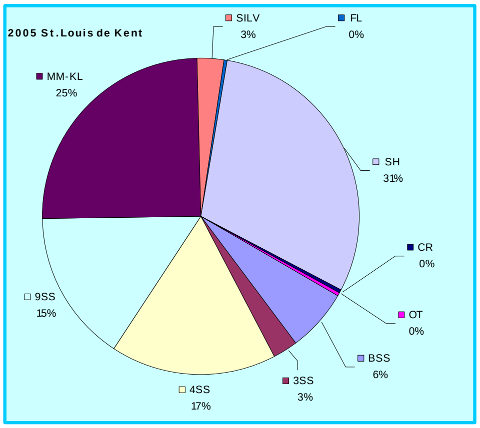


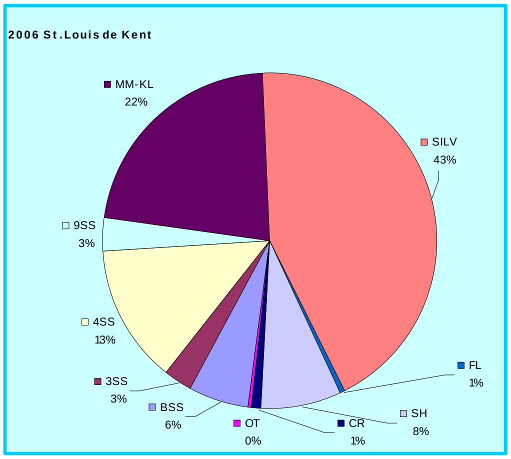
New Brunswick / St. Louis de Kent

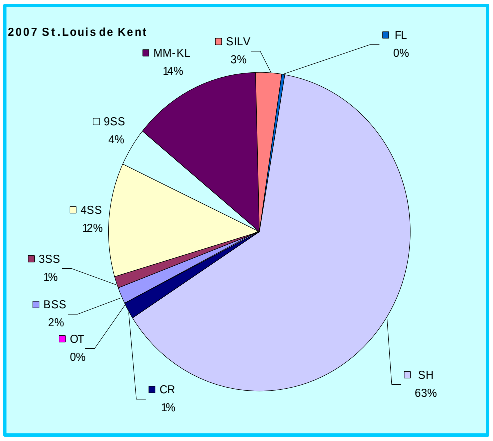
Black Spot Stickleback (BSS)
3 Spine Stickleback (3SS)
4 Spine Stickleback (4SS)
9 Spine Stickleback (9SS)
Mummichog/Killifish (MM-KL)

Atlantic Silverside (SILV)
Flounder Species (FL)
Shrimp Species (SH)
Crab Species (CR)
Other Species (OT)



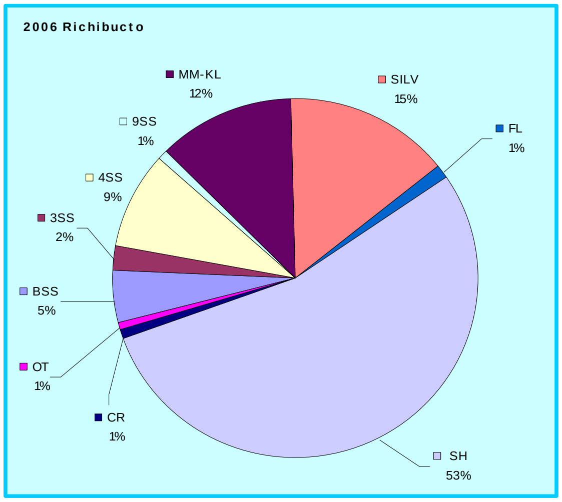


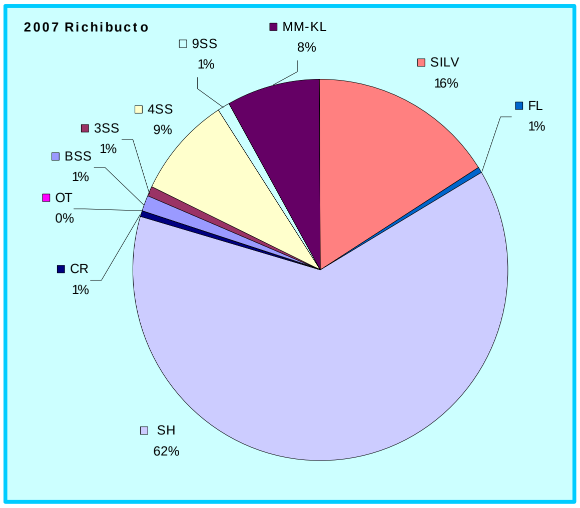




Richibucto / New Brunswick

Black Spot Stickleback (BSS)
3 Spine Stickleback (3SS)
4 Spine Stickleback (4SS)
9 Spine Stickleback (9SS)
Mummichog/Killifish (MM-KL)
Atlantic Silverside (SILV)
Flounder Species (FL)
Shrimp Species (SH)
Crab Species (CR)
Other Species (OT)

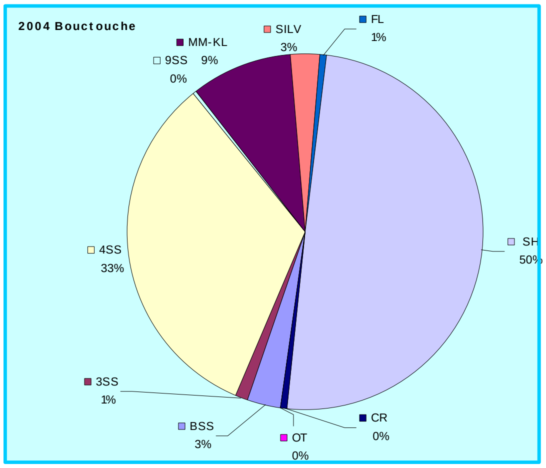


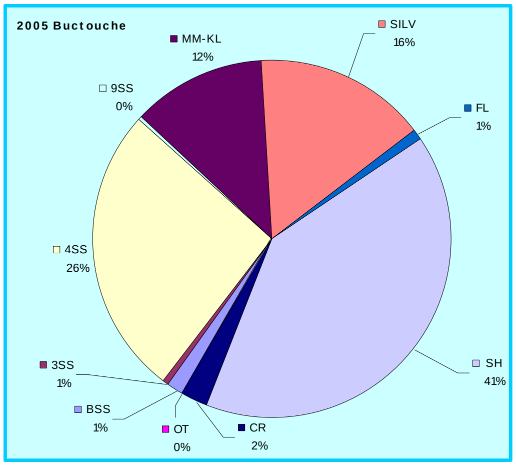


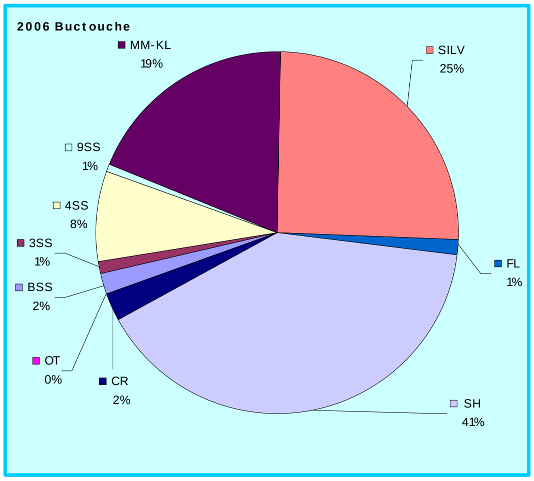
Bouctouche / New Brunswick

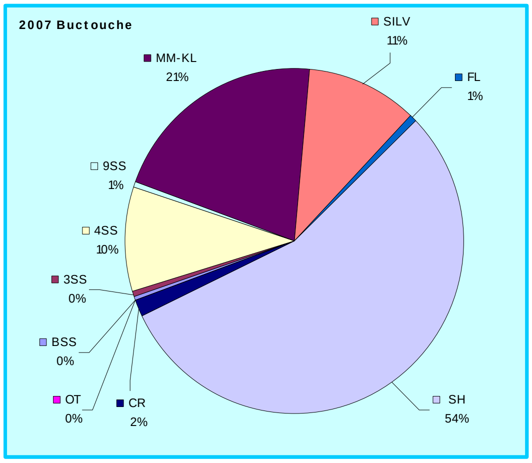
Black Spot Stickleback (BSS)
3 Spine Stickleback (3SS)
4 Spine Stickleback (4SS)
9 Spine Stickleback (9SS)
Mummichog/Killifish (MM-KL)

Atlantic Silverside (SILV)
Flounder Species (FL)
Shrimp Species (SH)
Crab Species (CR)
Other Species (OT)





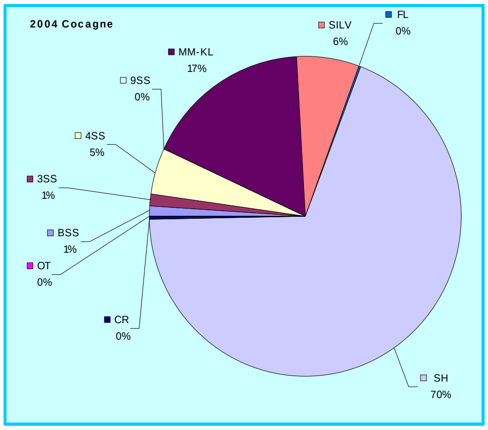


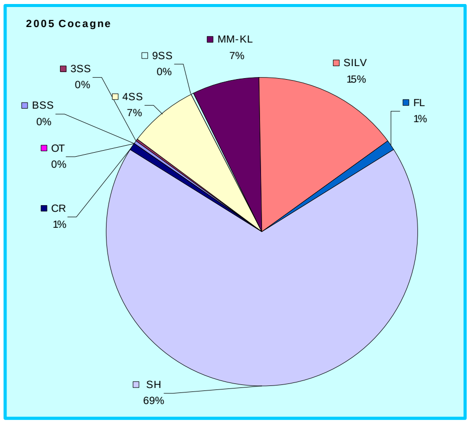


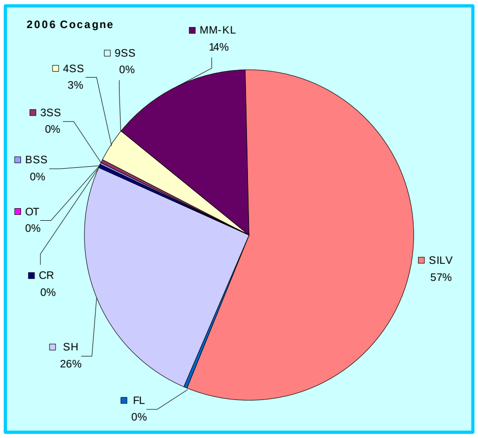
Cocagne / New Brunswick

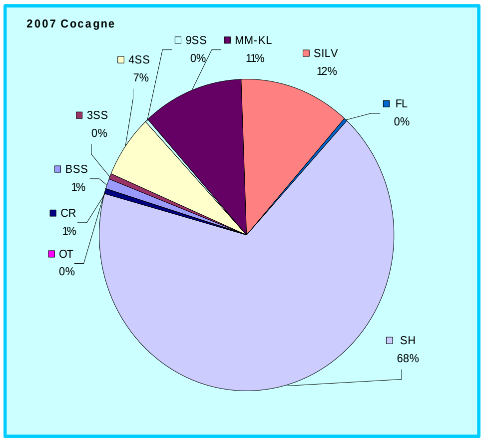
Black Spot Stickleback (BSS)
3 Spine Stickleback (3SS)
4 Spine Stickleback (4SS)
9 Spine Stickleback (9SS)
Mummichog/Killifish (MM-KL)

Atlantic Silverside (SILV)
Flounder Species (FL)
Shrimp Species (SH)
Crab Species (CR)
Other Species (OT)





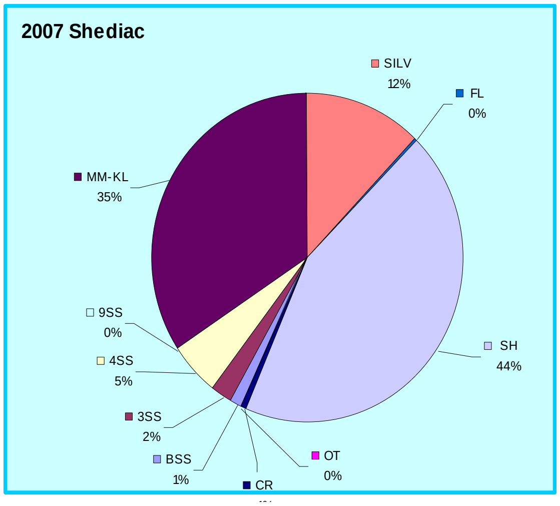




Shediac River / New Brunswick

Black Spot Stickleback (BSS)
3 Spine Stickleback (3SS)
4 Spine Stickleback (4SS)
9 Spine Stickleback (9SS)
Mummichog/Killifish (MM-KL)

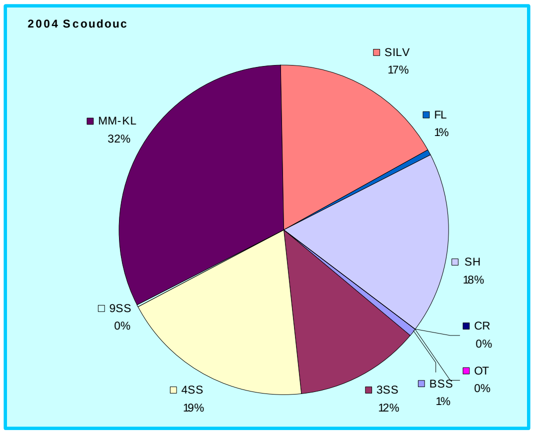
Atlantic Silverside (SILV)
Flounder Species (FL)
Shrimp Species (SH)
Crab Species (CR)
Other Species (OT)

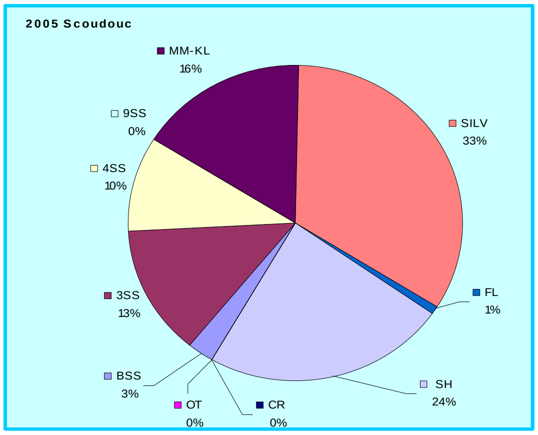


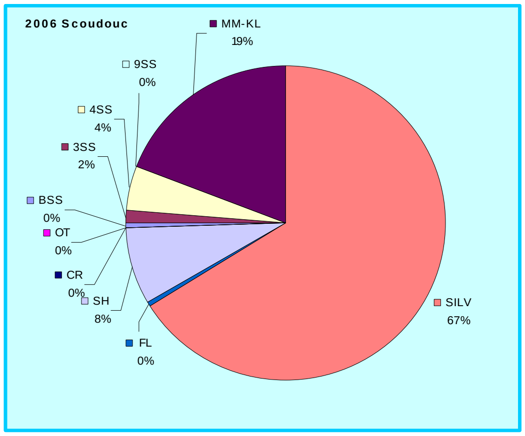
Scoudouc River/ New Brunswick (in Shediac)

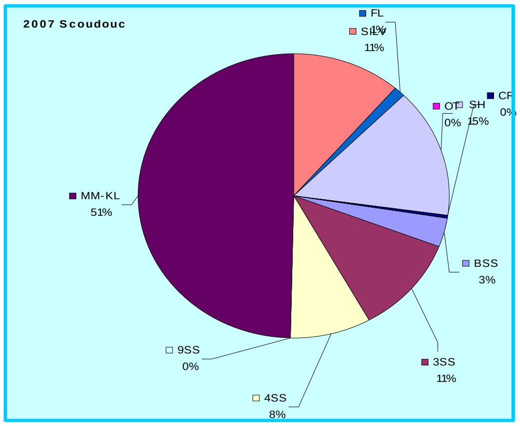
Black Spot Stickleback (BSS)
3 Spine Stickleback (3SS)
4 Spine Stickleback (4SS)
9 Spine Stickleback (9SS)
Mummichog/Killifish (MM-KL)

Atlantic Silverside (SILV)
Flounder Species (FL)
Shrimp Species (SH)
Crab Species (CR)
Other Species (OT)





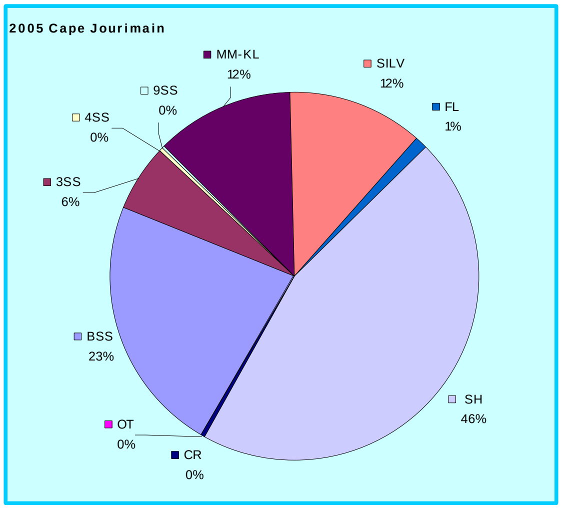


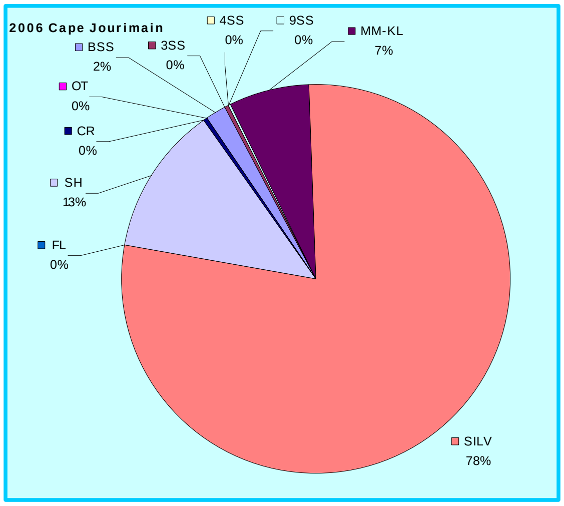


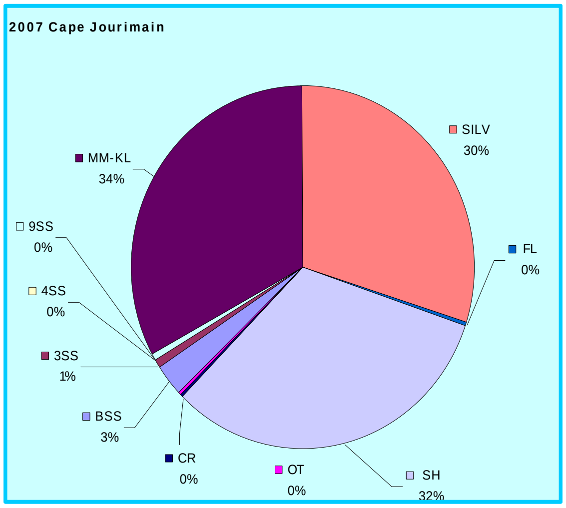
Cape Jourimain / New Brunswick

Black Spot Stickleback (BSS)
3 Spine Stickleback (3SS)
4 Spine Stickleback (4SS)
9 Spine Stickleback (9SS)
Mummichog/Killifish (MM-KL)

Atlantic Silverside (SILV)
Flounder Species (FL)
Shrimp Species (SH)
Crab Species (CR)
Other Species (OT)

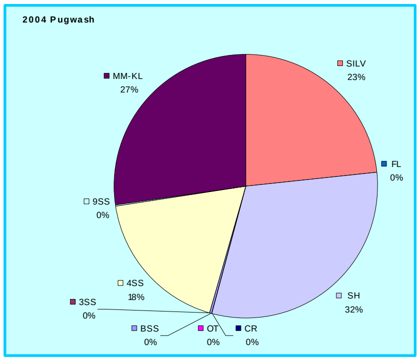


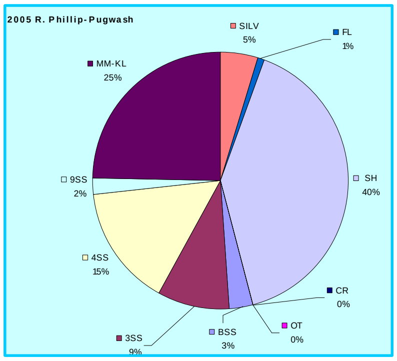


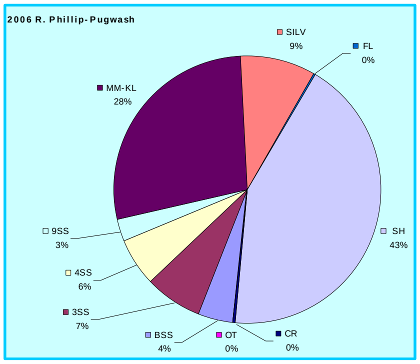


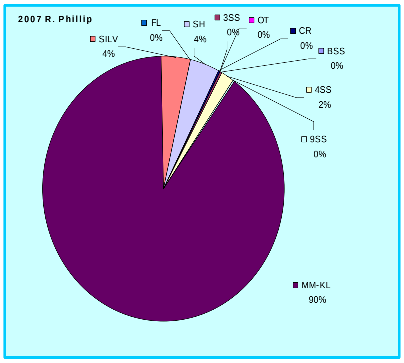
River Philip - Pugwash / Nova Scotia
(2004-2006)
River Philip 2007

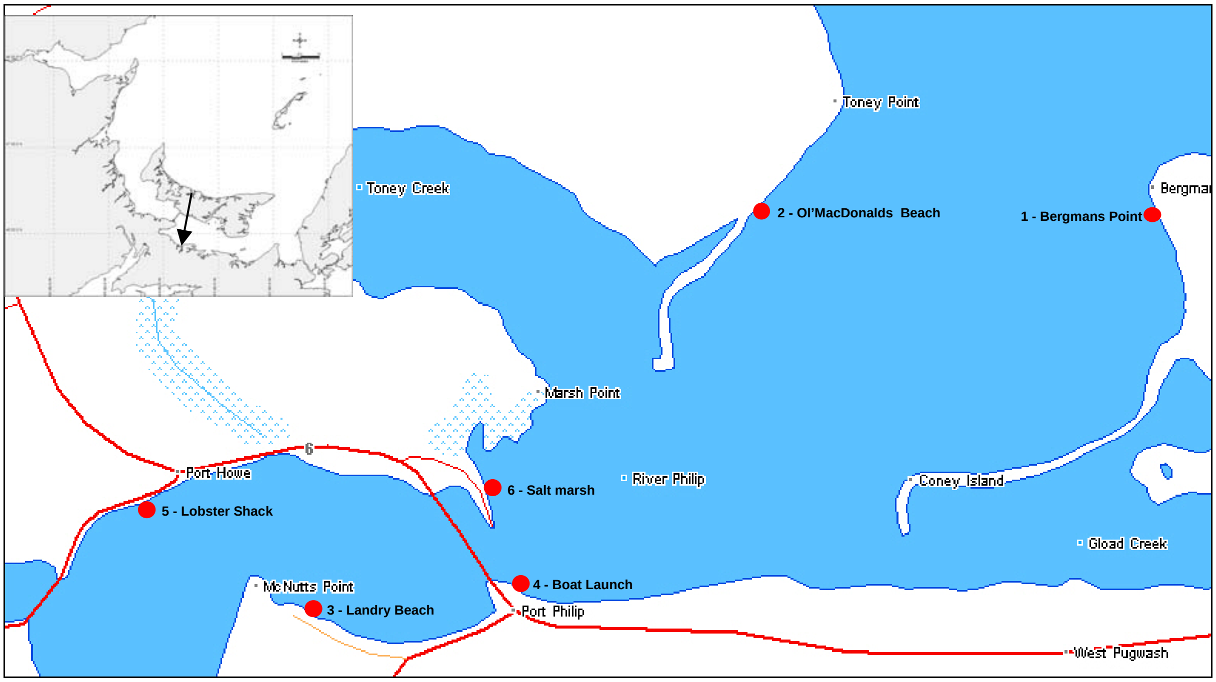
Black Spot Stickleback (BSS)
3 Spine Stickleback (3SS)
4 Spine Stickleback (4SS)
9 Spine Stickleback (9SS)
Mummichog/Killifish (MM-KL)
Atlantic Silverside (SILV)
Flounder Species (FL)
Shrimp Species (SH)
Crab Species (CR)
Other Species (OT)









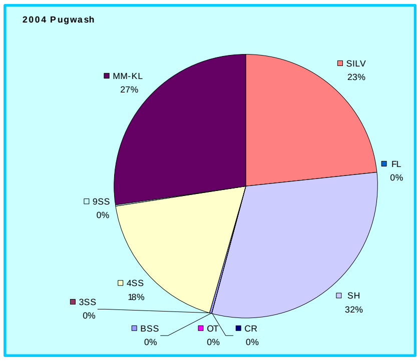


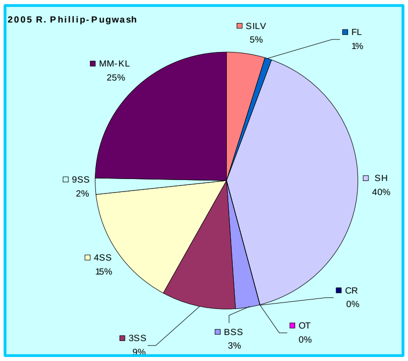
River Philip - Pugwash / Nova Scotia (2004-2006)

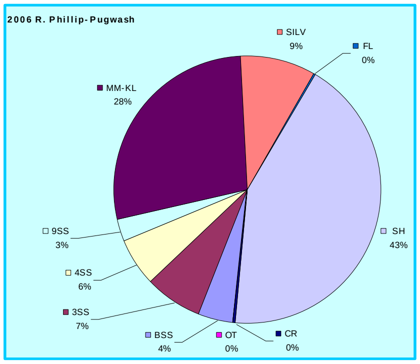
Pugwash 2007

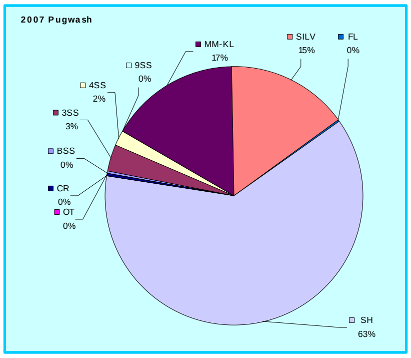
Black Spot Stickleback (BSS)
3 Spine Stickleback (3SS)
4 Spine Stickleback (4SS)
9 Spine Stickleback (9SS)
Mummichog/Killifish (MM-KL)

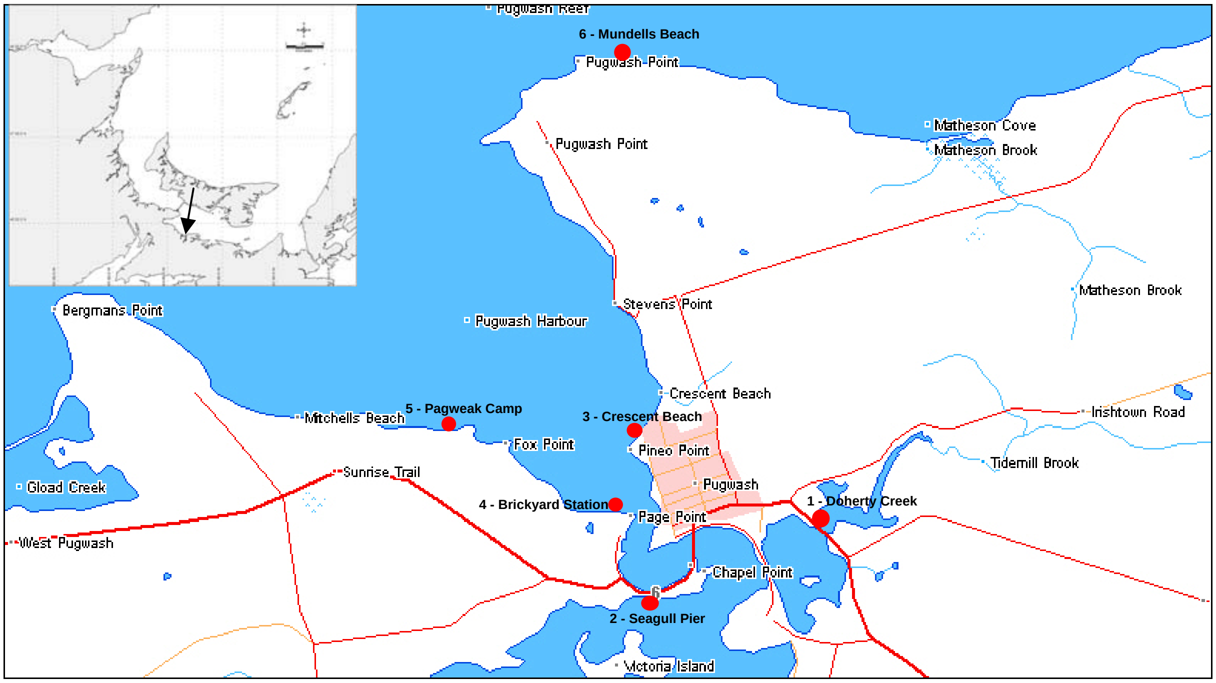
Atlantic Silverside (SILV)
Flounder Species (FL)
Shrimp Species (SH)
Crab Species (CR)
Other Species (OT)







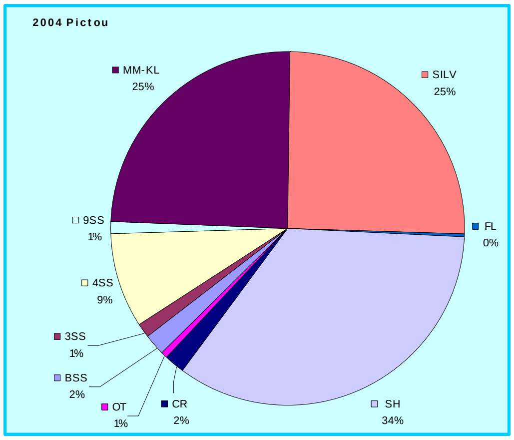


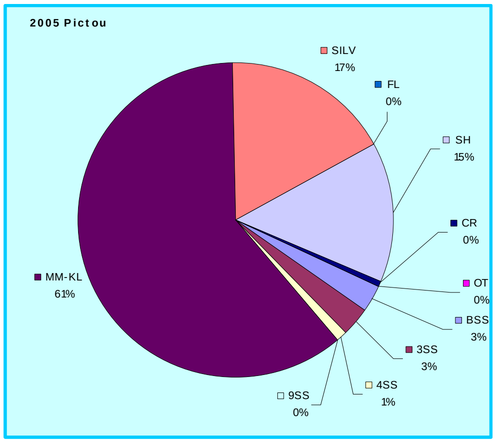


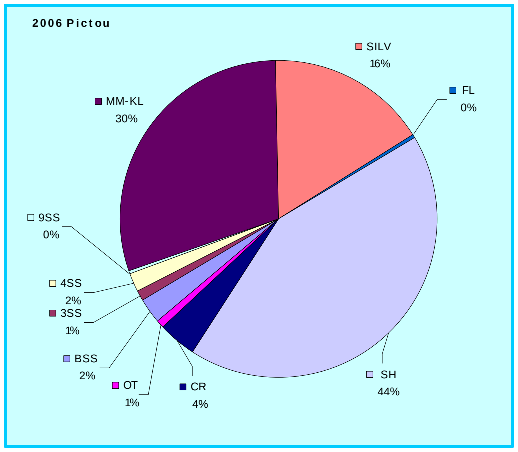
Pictou / Nova Scotia

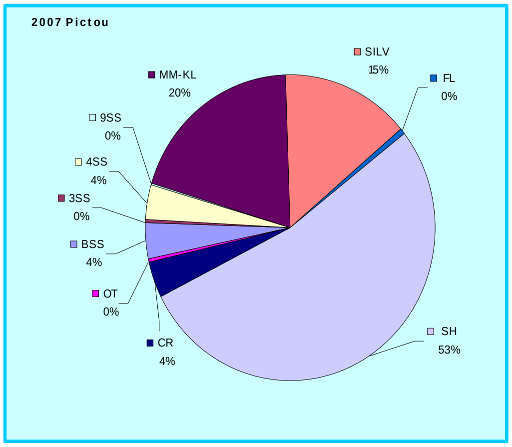
Black Spot Stickleback (BSS)
3 Spine Stickleback (3SS)
4 Spine Stickleback (4SS)
9 Spine Stickleback (9SS)
Mummichog/Killifish (MM-KL)

Atlantic Silverside (SILV)
Flounder Species (FL)
Shrimp Species (SH)
Crab Species (CR)
Other Species (OT)





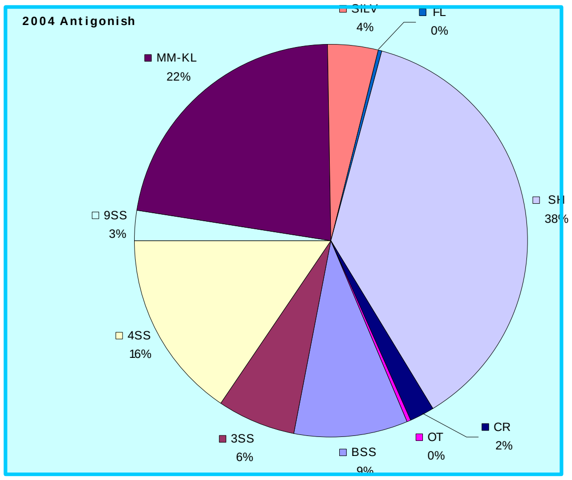


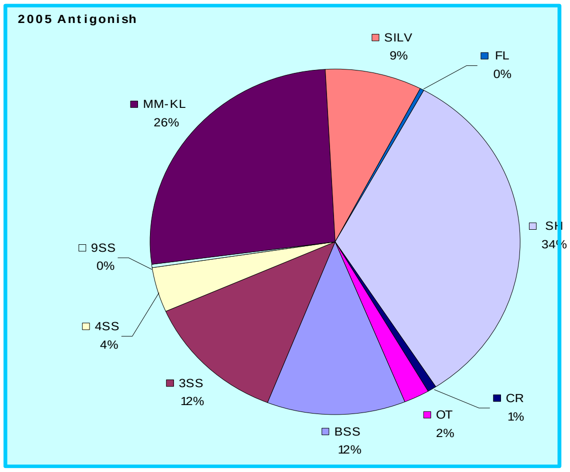


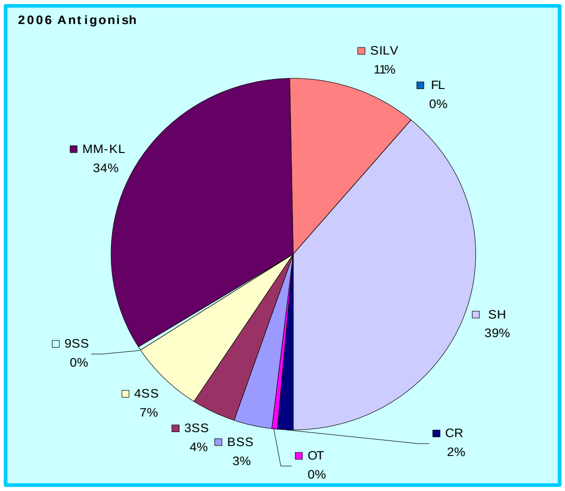
Antigonish / Nova Scotia

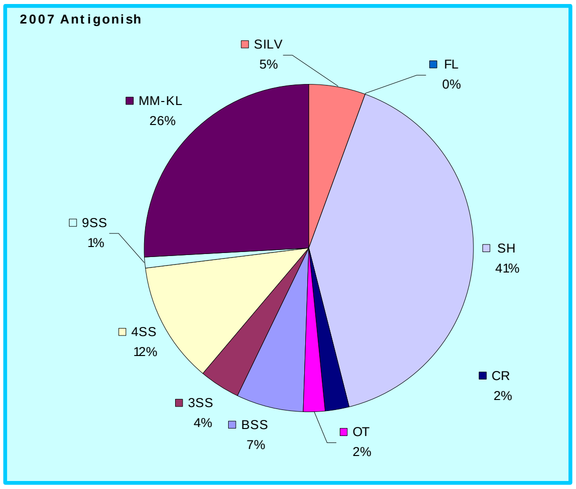
Black Spot Stickleback (BSS)
3 Spine Stickleback (3SS)
4 Spine Stickleback (4SS)
9 Spine Stickleback (9SS)
Mummichog/Killifish (MM-KL)

Atlantic Silverside (SILV)
Flounder Species (FL)
Shrimp Species (SH)
Crab Species (CR)
Other Species (OT)



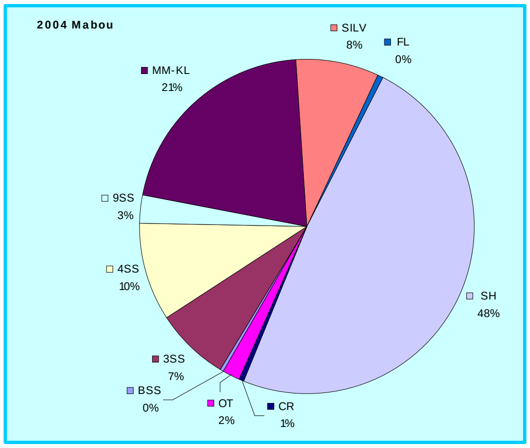


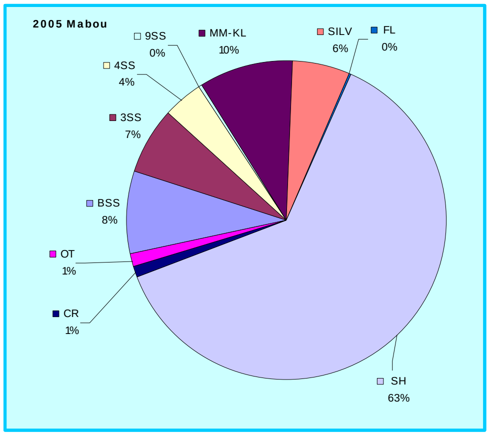


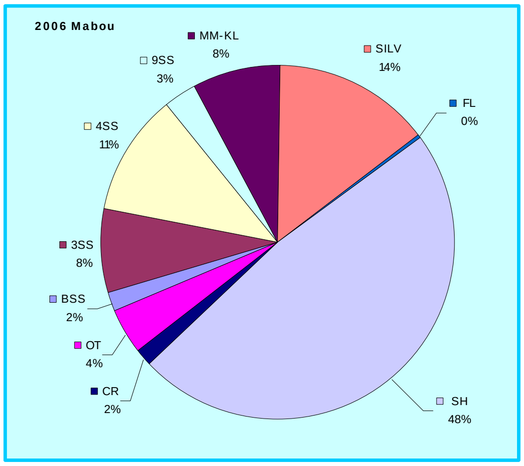


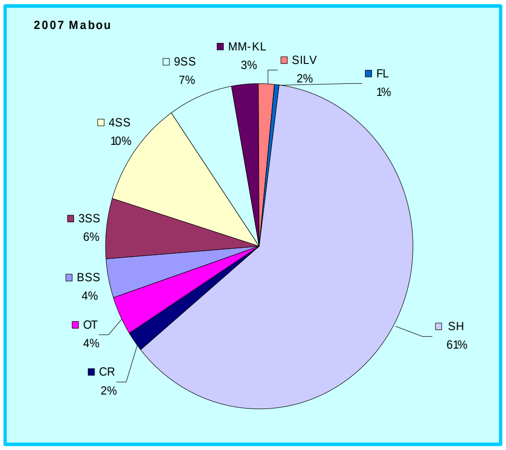
Mabou / Nova Scotia

Black Spot Stickleback (BSS)
3 Spine Stickleback (3SS)
4 Spine Stickleback (4SS)
9 Spine Stickleback (9SS)
Mummichog/Killifish (MM-KL)
Atlantic Silverside (SILV)
Flounder Species (FL)
Shrimp Species (SH)
Crab Species (CR)
Other Species (OT)





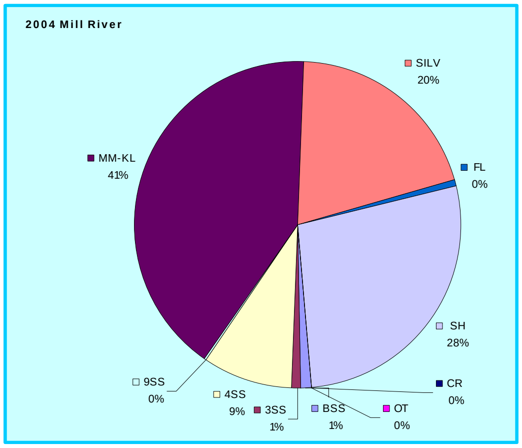


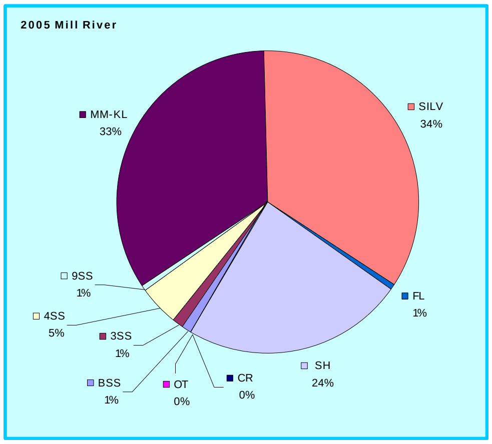


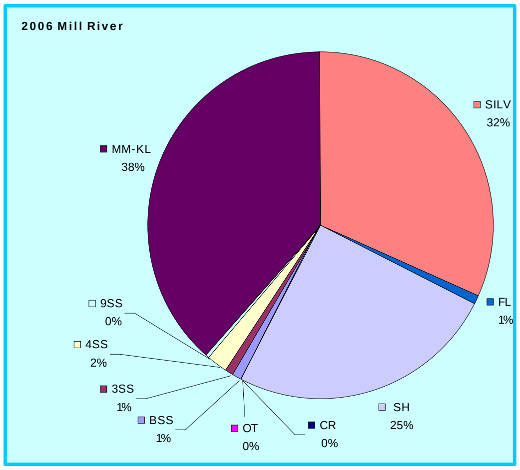
Mill River / Prince Edward Island

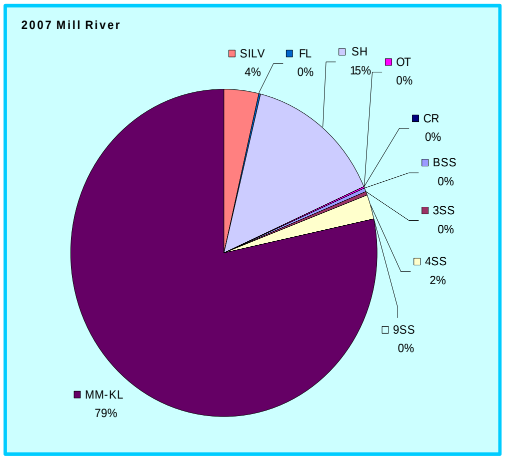
Black Spot Stickleback (BSS)
3 Spine Stickleback (3SS)
4 Spine Stickleback (4SS)
9 Spine Stickleback (9SS)
Mummichog/Killifish (MM-KL)

Atlantic Silverside (SILV)
Flounder Species (FL)
Shrimp Species (SH)
Crab Species (CR)
Other Species (OT)





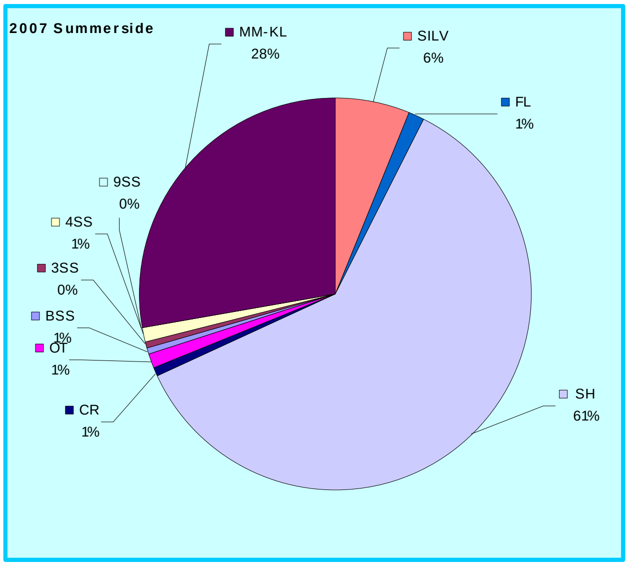


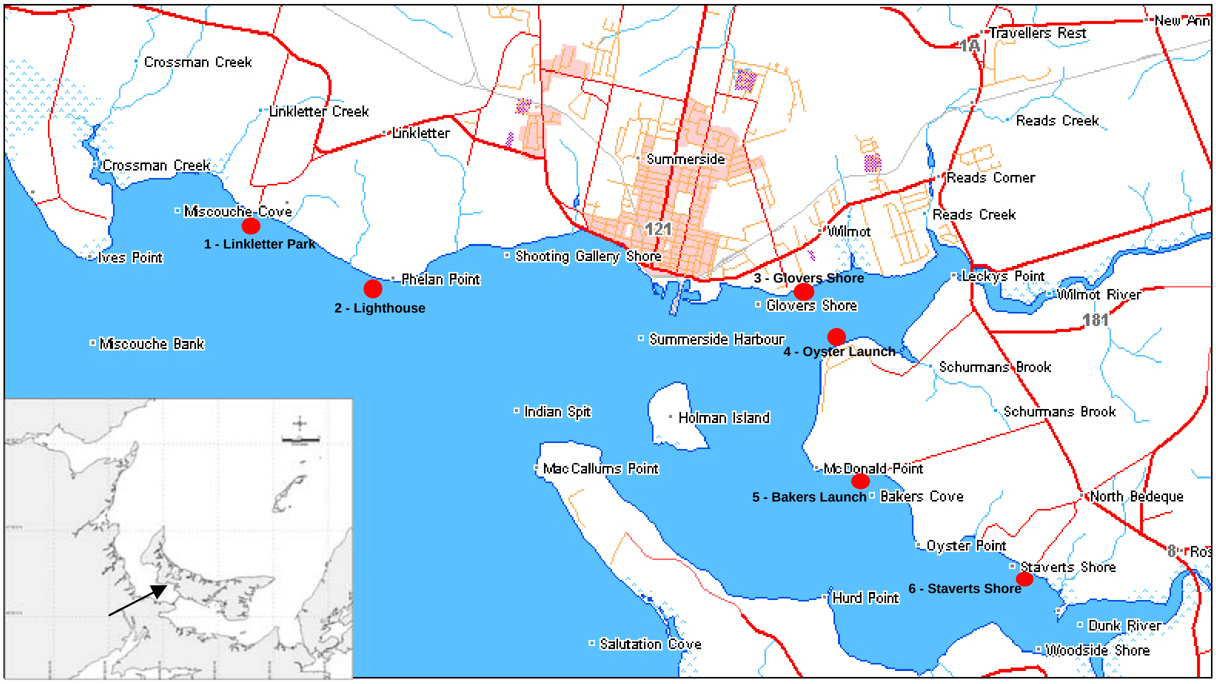


Summerside / Prince Edward Island

Black Spot Stickleback (BSS)
3 Spine Stickleback (3SS)
4 Spine Stickleback (4SS)
9 Spine Stickleback (9SS)
Mummichog/Killifish (MM-KL)

Atlantic Silverside (SILV)
Flounder Species (FL)
Shrimp Species (SH)
Crab Species (CR)
Other Species (OT)

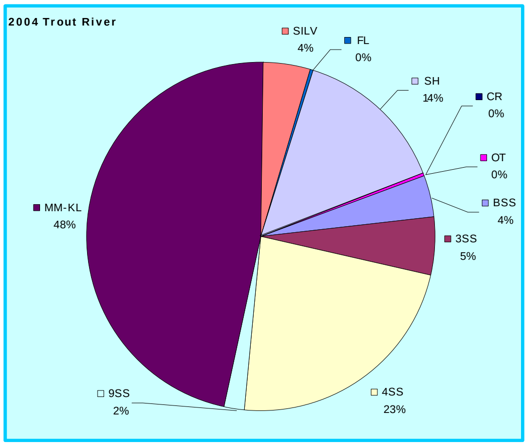


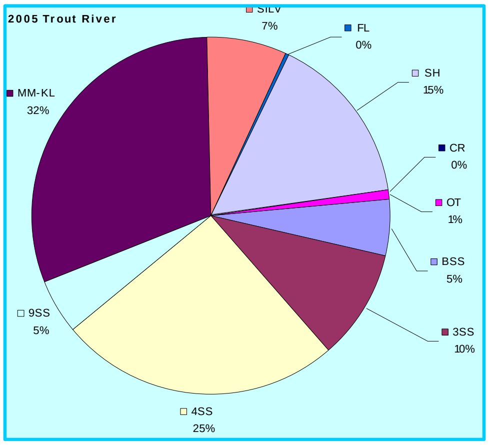


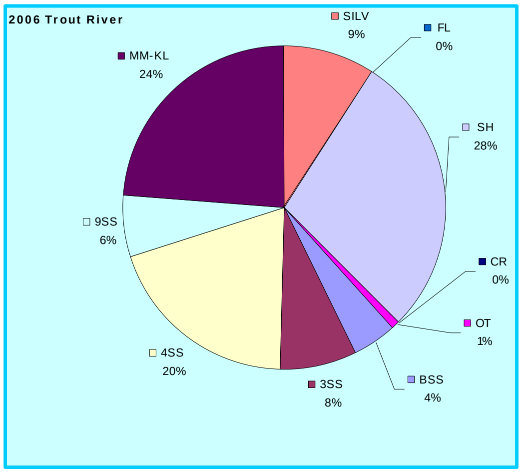
Trout River / Prince Edward Island

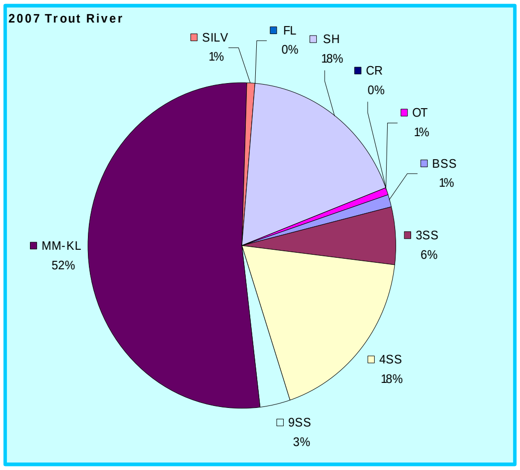
Black Spot Stickleback (BSS)
3 Spine Stickleback (3SS)
4 Spine Stickleback (4SS)
9 Spine Stickleback (9SS)
Mummichog/Killifish (MM-KL)

Atlantic Silverside (SILV)
Flounder Species (FL)
Shrimp Species (SH)
Crab Species (CR)
Other Species (OT)



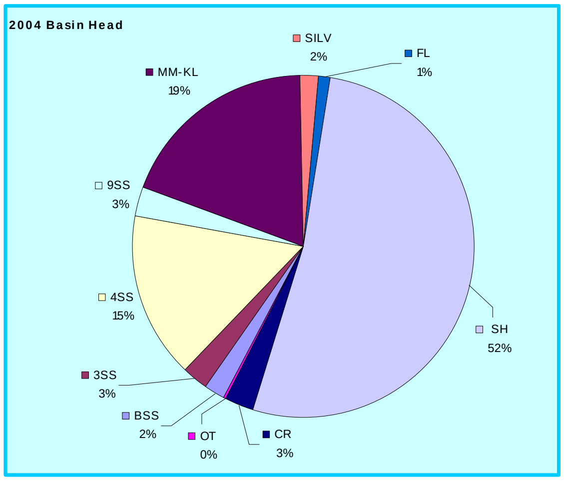


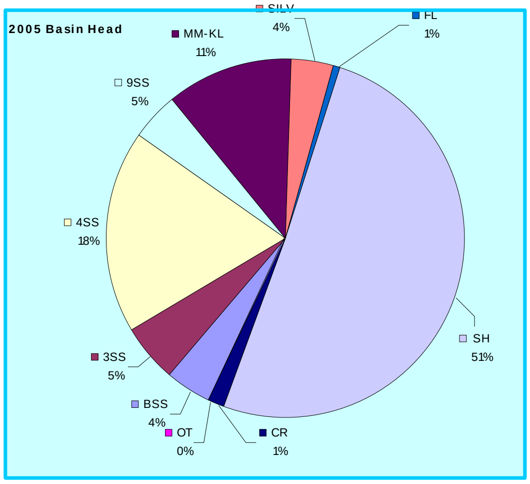


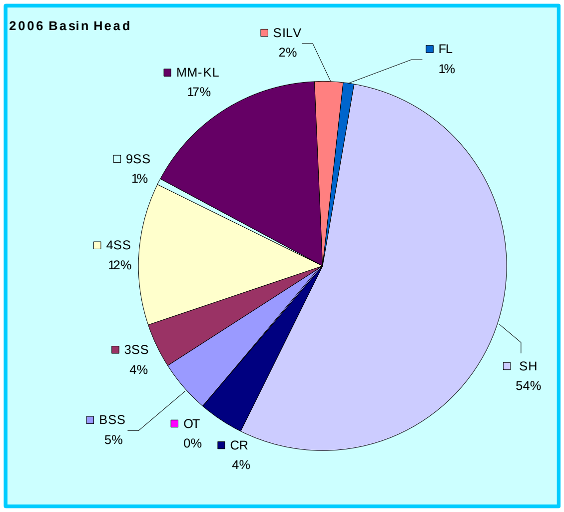


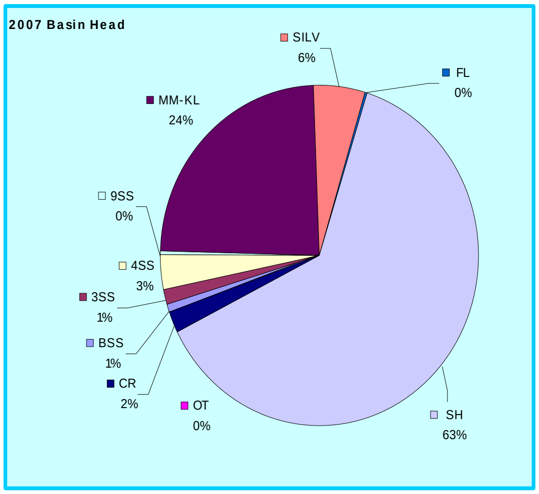
Prince Edward Island / Basin Head

Black Spot Stickleback (BSS)	Atlantic Silverside (SILV)
3 Spine Stickleback (3SS)	Flounder Species (FL)
4 Spine Stickleback (4SS)	Shrimp Species (SH)
9 Spine Stickleback (9SS)	Crab Species (CR)
Mummichog/Killifish (MM-KL)	Other Species (OT)





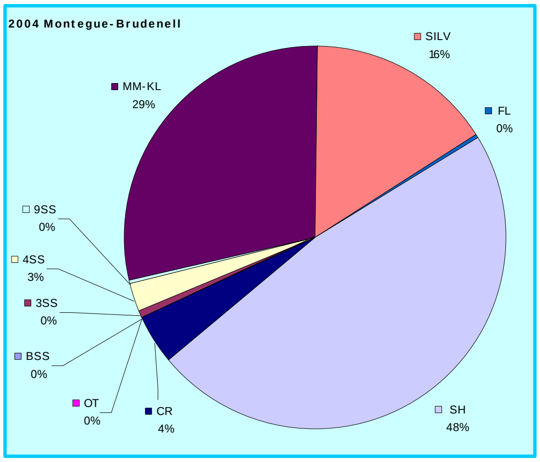


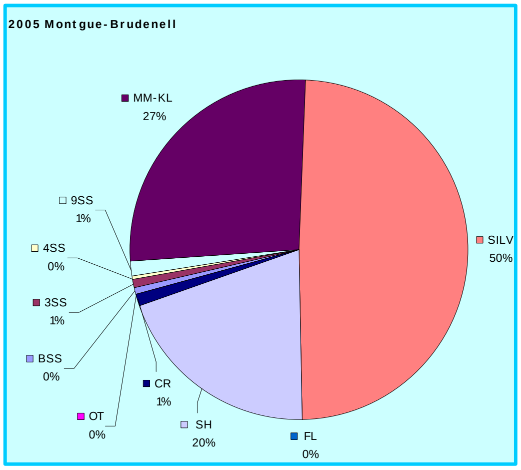


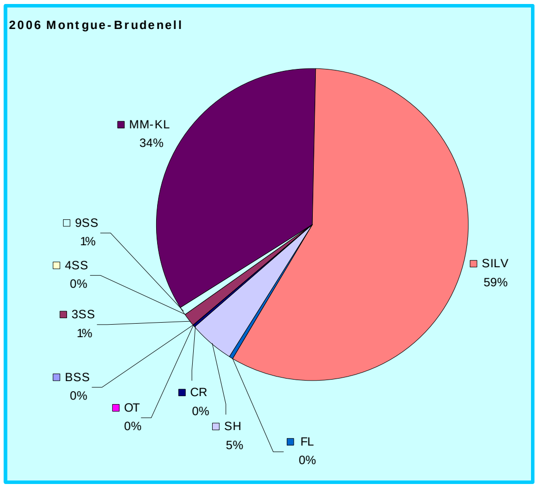
Prince Edward Island / Montague-Brudenell

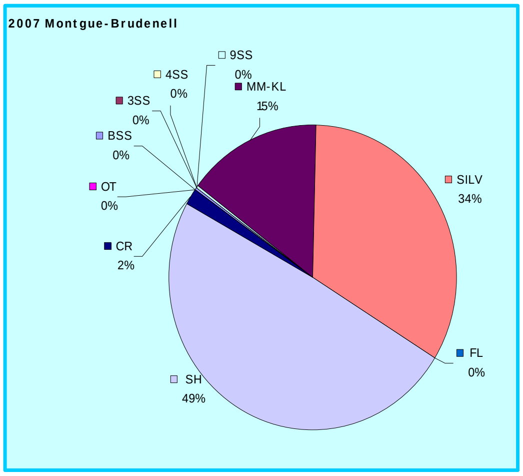
Black Spot Stickleback (BSS)
3 Spine Stickleback (3SS)
4 Spine Stickleback (4SS)
9 Spine Stickleback (9SS)
Mummichog/Killifish (MM-KL)

Atlantic Silverside (SILV)
Flounder Species (FL)
Shrimp Species (SH)
Crab Species (CR)
Other Species (OT)





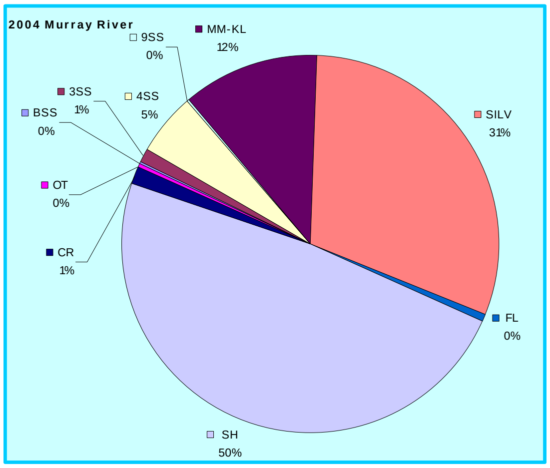


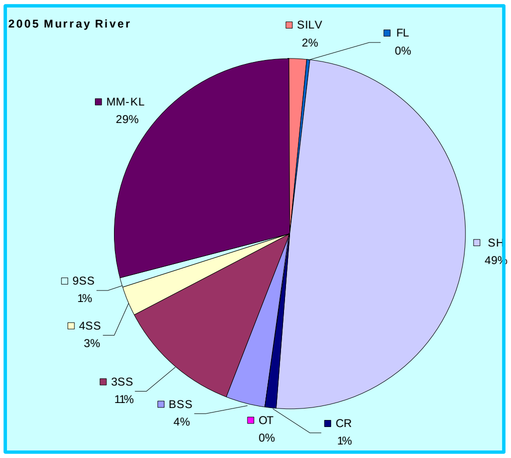


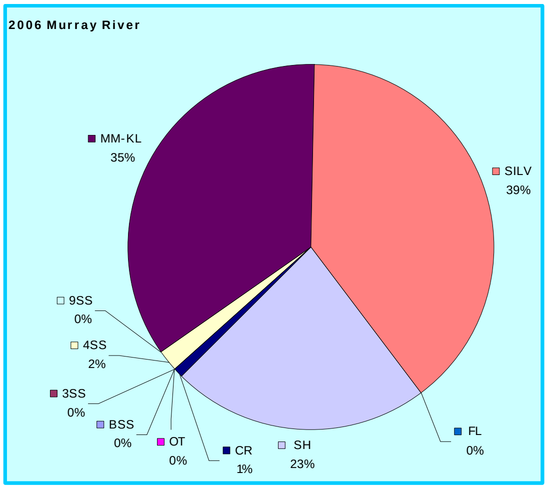
Prince Edward Island / Murray River

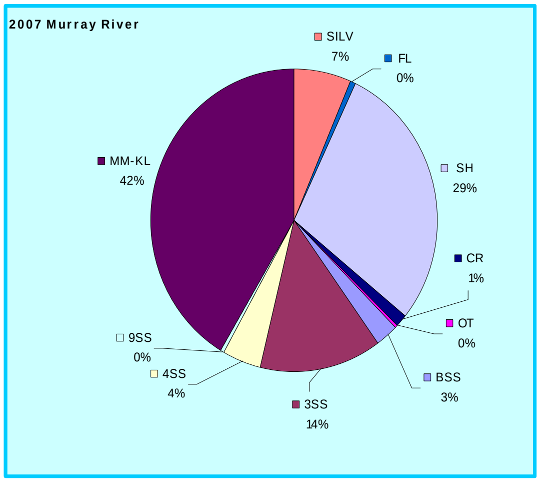
Black Spot Stickleback (BSS)
3 Spine Stickleback (3SS)
4 Spine Stickleback (4SS)
9 Spine Stickleback (9SS)
Mummichog/Killifish (MM-KL)

Atlantic Silverside (SILV)
Flounder Species (FL)
Shrimp Species (SH)
Crab Species (CR)
Other Species (OT)


2004 Murray River


2005 Murray River

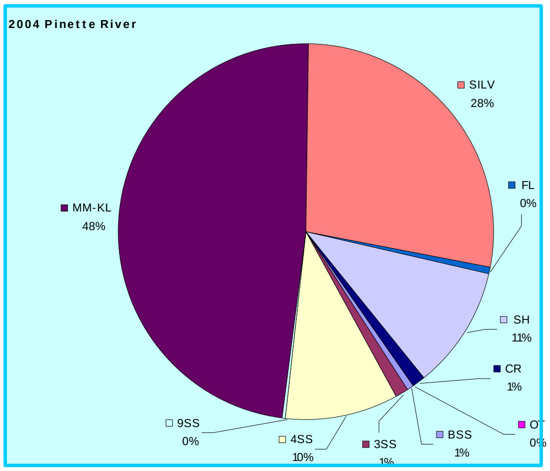

2006 Murray River

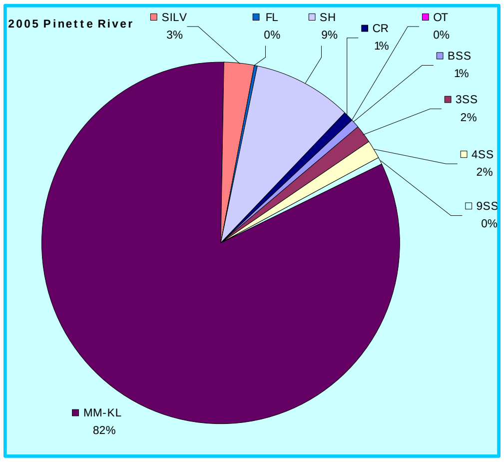

2007 Murray River

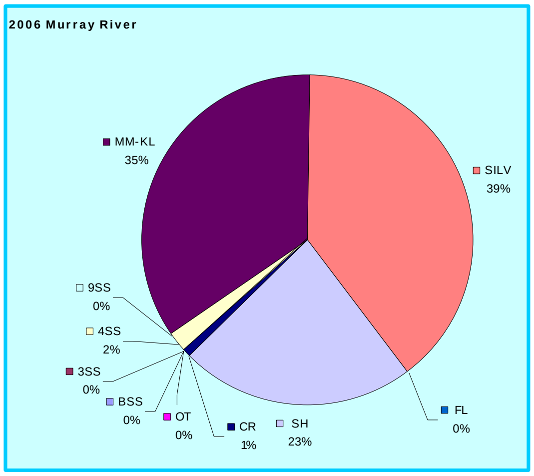
Prince Edward Island / Pinette River

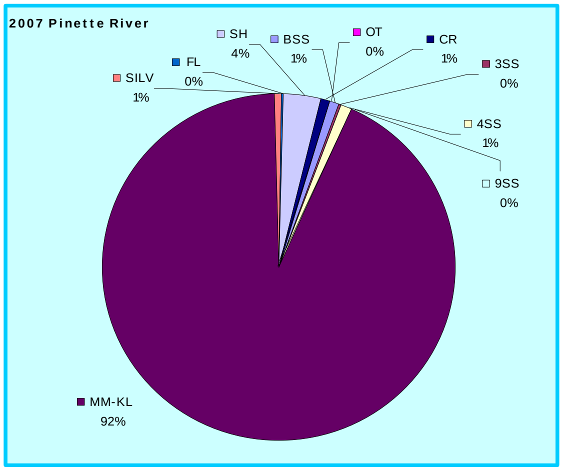
Black Spot Stickleback (BSS)
3 Spine Stickleback (3SS)
4 Spine Stickleback (4SS)
9 Spine Stickleback (9SS)
Mummichog/Killifish (MM-KL)

Atlantic Silverside (SILV)
Flounder Species (FL)
Shrimp Species (SH)
Crab Species (CR)
Other Species (OT)









From: Moore, Harrison M <Harrison.Moore@novascotia.ca>
Sent: April 7, 2021 4:46 PM
To: Clancey, Lewis <Lewis.Clancey@novascotia.ca>
Cc: Spencer, Amanda L <Amanda.Spencer@novascotia.ca>
Subject: RE: Town Point Oysters, Antigonish lease application information.

Thanks Lewis,

I will have a review of these as soon as I can. This is a very busy time of year and there appears to be a substantial amount of info.

Is this info part of any formal process? What is the next step (i.e. is the dept. supposed to respond to this info or is it info only?)

Thanks,

H

From: Boudreau, Louise O <Louise.Boudreau@novascotia.ca>
Sent: June 1, 2021 12:12 PM
To: Clancey, Lewis <Lewis.Clancey@novascotia.ca>; Feindel, Nathaniel J <Nathaniel.Feindel@novascotia.ca>; Ceschiutti, Robert <Robert.Ceschiutti@novascotia.ca>; Greenwood, Megan N <Megan.Greenwood@novascotia.ca>
Cc: Moore, Harrison M <Harrison.Moore@novascotia.ca>; O'Brien-Latham, Lesley <Lesley.OBrien-Latham@novascotia.ca>
Subject: L&F review of new AQ 1442, 1443, 1444 submission

Hello Everyone,

Thank you for providing Department of Lands and Forestry with a new submission to address biology concerns for AQ 1442, 1443, 1444. Our biology reviewer was pleased with the work that went into this submission. We appreciate you working with the proponent on this package to address our concerns. The new information meets our requirements:

> ***Update May 27, 2021:***
>
> *In response to L&F's request for additional information in its original review in dated March 26, 2021, and following review of the additional information provided, Lands and Forestry is satisfied that the proposed oyster lease will not pose undue negative impact to the ecosystem. The additional information thoroughly outlines the potential impacts to the local ecosystem including eelgrass. See below for detailed comments.*

Please see the Departmental comments (attached). The assessment of new information appears in blue italic font.

Warm Regards,

Louise

Louise Boudreau

Policy Analyst

Department of Lands and Forestry

Founders Square |1701 Hollis Street, 3rd Floor | Halifax, NS B3J 2T9 |

424-3530

Updated Response NS Lands and Forestry May 27, 2021

Network Agency Review of an Aquaculture Application

Agency	**Lands and Forestry**
Division (if applicable)	**Policy Division on behalf of the Department Lands and Forestry**
Date	**Dec. 7th 2020**
File No.	**AQ 1442 1443 1444**
Type of application	**New Marine Shellfish-**
Information Provided	**Subsequent biology package submitted April 2021**

Please provide comments, concerns, recommendations, or requirements on the above stated application for a marine aquaculture license. Please include the criterion /criteria within your jurisdiction or mandate that your feedback is based upon. Similarly, if additional information is required to make a determination, please include the criterion /criteria within your jurisdiction or mandate that your request is based upon.

☐ No concerns regarding the proposed development
☐ Concerns with development are expressed below
☐ Request modifications to the proposed development (described below)
☐ Required or recommended conditions (described below)
☐ Request additional information (described below)
☐ Request meeting with applicant and NSDFA (described below)
☐ No comments on the application

Comments, concerns, recommendations, and/or required conditions including the criterion /criteria within your jurisdiction or mandate that your feedback is based upon. (Attach comments if preferred, or add additional pages, as required.):

The Department of Lands and Forestry has the following comments:

UPDATE May 27, 2021: On April 12th, 2021 Town Point Consulting (TPC) provided a response (through the Department of Fisheries and Aquaculture- NSDFA) to Lands and Forestry's comments/concerns entitled: "Response re NSDLAS concerns March 26, 2021". NSDFA provided a series of PDF documents to support this document with scientific literature. This information was reviewed, and comments are provided below in *italics* in response to each of Lands and Forestry's initial concerns.

Crown Land:

According to the records on file at the Crown Land Information Management Centre, the subject area is considered ungranted Crown land with no encumbrances.

Parks and Beaches Act:

The Department has the following concerns:

1. The proposed lease is adjacent to a pending provincial park: Dunns Provincial Park Reserve. Recreation is a mandate of the Department under the Parks and Beaches Act. The area is known for recreational activities such as boating, fishing, and kayaking. Meaningful consultation with recreational users to better understand recreational values needs to take place to address concerns and mitigate impacts.

Wildlife/Wildlife Habitat:

Request for Additional Information:

The Department does not have sufficient information to complete a fulsome review of this application. The Department has concerns regarding the lack of evidence in the document including but not limited to baseline data, scientific references or evidence, and lack of mitigation for wildlife issues. There is no evidence that this project will not contribute to negative impacts to the local ecosystem and wildlife. There are only a few instances of the proponent outlining a possible negative impact of the project and providing mitigation. More information is needed to determine if there will be more negative impacts resulting from this project and if more mitigation is needed.

Update May 27, 2021: *In response to L&F's request for additional information in its original review in dated March 26, 2021, and following review of the additional information provided, Lands and Forestry is satisfied that the proposed oyster lease will not pose undue negative impact to the ecosystem. The additional information thoroughly outlines the potential impacts to the local ecosystem including eelgrass. See below for detailed comments.*

Comments on the Application Document

1. The Department is concerned that the application document indicates that there will be some changes to the local environment (negative or positive) but does not clearly identify those changes. It is also a concern that potential negative impacts are not considered or dismissed and that no mitigation is put forward for these potential impacts.

   UPDATE May 27, 2021: *Lands and Forestry (L&F) response- Upon receipt and review of the information provided by both NSDFA and TPC, Lands and Forestry has sufficient evidence to determine that the wildlife/habitat concerns previously raised have been addressed to support the application. The data package provides both the potential impacts and mitigation to ensure no undue ecosystem level impacts will occur. The proposed development incorporates appropriate mitigations (i.e. spacing, stocking density etc.).*

2. There is an overall lack of baseline data which is necessary to monitor the project going forward. Since this project does not provide baseline data, mitigation, or data to support the safety of this project, it can not be determined if this oyster farm could cause harm to Species at Risk and other species protected through the *NS Wildlife Act* and the *Migratory Birds Convention Act*.

   Update May 27, 2021: *L&F response- Upon receipt and review of the information provided by both NSDFA and TPC, Lands and Forestry now has a clear understanding of the necessary monitoring procedures to be put in place along with the potential risks. Because of the overall low impact of the project and the monitoring that is to be conducted, Lands and Forestry is satisfied with the proposed mitigation and baseline data collected to date.*

3. The document relies on repetitious data, individual opinions, and anecdotes. Rather, it relies on assertions made by individuals, many whose names are redacted and without supporting evidence. The document provides almost no evidence for its assertions on wildlife. The Piping Plover report is an exception and provides adequate detail on the species and how mitigation was incorporated. The overall quality of the document does not meet the information needs of the Department, and lack structure making it difficult to navigate (no table of contents, page numbers, references, many repetitions, and no background data/preamble). There are many instances indicating that literature reviews were completed but the report fails to provide evidence of this work.

   Update May 27, 2021: *L&F response- the response document provided by TPC was well organized, provided adequate support and proper references to address the environmental considerations of the project and outlined potential risks and mitigations. Upon review of the supporting documents, Lands and Forestry is satisfied that proper due diligence has been conducted by the proponent and management techniques that will lessen ecological impact have been/will be employed.*

4. The report is largely void of biological data, evidence, or baseline data (other than Piping Plover) necessary to assess the environmental impact.

   Update May 27, 2021: *L&F response- See above*

Recommendations

The Department does not have sufficient information to fully assess this proposal and provide recommendations for mitigation measures concerning the potential biodiversity impacts. Specifically, the Department requires:

1. Base line data and analysis that identifies the potential negative impacts on the ecosystem due to the increased load of shellfish. Ecosystem level impacts include negative impacts that could potentially harm the aquatic ecosystem and species (vegetation, benthic species etc.) and have consequent impacts on the food chain

2. ***Update May 27, 2021:*** *L&F response- the information package provided thoroughly outlines these potential impacts and provides sufficient evidence to conclude that the environmental impacts have been considered and mitigated. No ecosystem level/food chain impacts are anticipated.*

3. A comprehensive wildlife management plan that identifies impacts on birds is required. This was identified as a potential issue in the application. The plan must identify which species could be impacted, how these impacts can be mitigated and how to monitor for incidental impacts (entanglement, disturbance during resting periods, noise issues etc.). A thorough risk assessment is required to create this wildlife management plan.

   Update May 27, 2021: *L&F response- TPC and DFA have provided more information on the potential impacts to wildlife, including birds. Due to the nature of the project, it is not anticipated that there will be undue negative effects to avifauna. Furthermore, the proponent has outlined how they will conduct operations to further reduce any human wildlife conflict.*

4. Baseline data for benthic invertebrates, vegetation, water quality etc. This is necessary to ensure the long-term sustainability of the project and harbour ecosystem. This data should be collected, and a monitoring plan should be created to adequately monitor environmental impacts; positive or negative.

   Update May 27, 2021: *L&F response- A monitoring plan is in place that provides for partnering with NSDFA, DFO and other non-government agencies. This monitoring is acceptable to Lands and Forestry and the department will defer aquatic monitoring to DFO.*

5. Information (literature review, pre-disturbance data etc.) collected or collated on the following issues:
   a. Effects of de-fouling on the local water quality, species etc.
   b. Noise from farm- effects on birds and any other potentially impacted wildlife (A literature review may suffice).

c. Impacts for birds etc. where potential impacts may be caused by suspended oysters, noise, and the effects bird waste has on oyster quality if nearby.
d. Possible effects on eelgrass and baseline data on current extent of eelgrass.

Update May 27, 2021: *L&F response- The potential effects and monitoring of impact to eelgrass beds have been thoroughly discussed. The proponent will continue to monitor any impacts to eel grass and has adopted management techniques to reduce these impacts. Water quality issues, if any, are deemed to be localized, and are not anticipated to occur for this project. The proposed lease occupies less than 2% of the harbour and due to natural and historical eutrophication of the harbour it is anticipated that the operation may increase habitat suitability. Furthermore, historic oyster concentration exceeds the levels in the proposal. The impacts to birds for this type of project is low as per the information included in the package. Noise from the operation may pose an issue but this is not well studied, and the proponent has outlined ways in which they have worked to lessen this potential impact.*

Lands and Forestry is satisfied with the proposal upon review of the additional information.

Recommended Conditions of Approval

The proponent must develop a wildlife management plan that is acceptable to the Department of Lands and Forestry. The proponent is required to implement the wildlife management plan as approved.

Update May 27, 2021: *L&F response- a wildlife management plan is no longer deemed necessary by Lands and Forestry as a condition after review of the additional data packages provided. Appropriate buffer/setback has been included to reduce disturbance to SAR species (i.e. Piping Plover).*

The proponent must:

1. Not disturb, harass, or chase congregated birds (waterfowl, geese, cormorants etc.), especially during January – March.
2. Not perform any work on Captains Island.
3. Remove any gear/equipment that is washed ashore/deposited on any Crown land at their expense.
4. Report all wildlife mortalities (entanglements etc.) to the Department of Lands and Forestry Regional Biologist and any other appropriate agency (if a marine mammal, Species at Risk etc.) within 5 business days.

Update May 27, 2021: *L&F response- TPC/the proponent has and must continue to include mitigation technics for all operations.*

Public Notice and Disclosure

As part of the process for deciding on an application, it may be necessary for the Nova Scotia Department of Fisheries and Aquaculture ("Fisheries and Aquaculture") to disclose the collected network review information to the applicant and other government bodies, including, if applicable, the Nova Scotia Aquaculture Review Board for use at an adjudicative hearing relating to the application in question.

In accordance with departmental policy, which seeks to promote public involvement in the process for deciding on aquaculture applications, Fisheries and Aquaculture will disclose aquaculture application information, including network review information, on the departmental website.

Privacy Statement

The network review information collected as part of an aquaculture application will only be used or disclosed by Fisheries and Aquaculture for the purpose of deciding on the application.

All application information collected is subject to the Freedom of Information and Protection of Privacy Act ("FOIPOP") and will only be used or disclosed in accordance with FOIPOP.

Spencer, Amanda L

From: Ernie Porter <ernieporter77@gmail.com>
Sent: June 3, 2021 11:35 AM
To: Clancey, Lewis
Cc: Spencer, Amanda L
Subject: Re: L&F review of new AQ 1442, 1443, 1444 submission
Attachments: LF-AQ#1442 1443 1444- v4 sent to AQ.docx

Hi Lew

This is great news! Thank you very much. Is there now enough in place for Nathaniel to begin

writing the affidavits? Best regards,
Ernie

Sent from my iPhone

> On Jun 3, 2021, at 11:05 AM, Clancey, Lewis
>
> <Lewis.Clancey@novascotia.ca> wrote: Hi Ernie,
>
> I'm please to provide you with this updated review (below and attached) from the Network Review Partners at NSL&F.
>
> Lew

APPENDIX J: OFFICE OF ABORIGINAL AFFAIRS (NOW OFFICE OF L'NU AFFAIRS)

From: Greenwood, Megan N
Sent: October 7, 2020 2:29 PM
To: Rillie, Claire Z <Claire.Rillie@novascotia.ca>; Marshman, Kendra Alair <Kendra.Marshman@novascotia.ca>
Cc: Ceschiutti, Robert <Robert.Ceschiutti@novascotia.ca>; King, Matthew S <Matthew.King@novascotia.ca>; Feindel, Nathaniel J <Nathaniel.Feindel@novascotia.ca>; Clancey, Lewis <Lewis.Clancey@novascotia.ca>
Subject: AQ#1442, 1443, 1444 - New Application

Attention: Office of Aboriginal Affairs

Please see the attached memo for three (3) Aquaculture Application Nos. 1442, 1443, 1444 in Antigonish Harbour, Antigonish County.

Please respond with your feedback by **November 9, 2020.**

The attached zipped shapefile requires specific geospatial software to open and is intended for convenient use by GIS professionals only.

Megan Greenwood

Licensing Coordinator

Nova Scotia Department of Fisheries and Aquaculture

1575 Lake Road

Shelburne, NS

B0T 1W0

Phone 902-875-7443

Fax 902-875-7429

From: Marshman, Kendra Alair
To: Greenwood, Megan N
Cc: Ceschiutti, Robert; King, Matthew S; Feindel, Nathaniel J; Clancey, Lewis; Rillie, Claire Z
Subject: RE: AQ#1442, 1443, 1444 - New Application
Date: November 6, 2020 4:56:09 PM
Attachments: 20 2020-08-06 Consultation Letter to Millbrook TEMPLATE.docx
20 2020-08-06 Consultation Letter to Sipekne"katik TEMPLATE.docx
20 2020-08-06 Consultation Letter to Assembly TEMPLATE.docx
05 2020-07-16 Consultation Correspondence Protocol and Contact List.docx

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Hello Megan,

Claire and I have screened the above-noted application for Aboriginal consultation purposes on behalf of OAA and we advise consultation at the moderate level with the Assembly of Nova Scotia Mi’kmaw Chiefs, Millbrook First Nation, and Sipekne’katik First Nation. Our rationale is as follows:

11) The proposed aquaculture leases are new applications for suspended culture of American oyster located in Antigonish Harbour where shellfish harvesting is known to occur;
12) The development plan submitted for these applications makes reference to an associated land-based nursery facility – No. 1422 - which was licenced by NSDFA on April 24, 2020;
13) A notification letter on Aquaculture Application No. 1422 was sent to Paqtnkek First Nation and copied to KMKNO on May 8, 2019. However, no response was received;
14) The proposed aquaculture lease will involve the use of submerged Crown land in Antigonish Harbour;
15) Possible environmental impacts from the construction of a new depuration facility;
16) Potential impacts on birds from the addition of suspended gear, including endangered Piping Plovers;
17) As identified by CCH, there are 3 archaeological sites within 1 km of the proposed site;
18) The proposed site is located approximately 30 km from Paqtnkek First Nation.

Please prepare an offer to consult letter for all 13 Chief and Councils using the templates attached. I’ve also attached our correspondence protocol to guide you when it comes time to send the letters out. OAA will populate the consultation screening section of your letter once you’ve prepared the first draft for me to review.

As I’ve mentioned before, given the capacity-related constraints due to COVID-19, OAA is recommending adding additional time to all consultation-related correspondence. We have also developed some standardized wording to accompany extensions, for example:

“We would like to hear from the Mi’kmaq of Nova Scotia about the details of any asserted Aboriginal or Treaty rights that could be adversely impacted by this particular project/initiative. We would appreciate a response concerning this project/initiative by [date]. We have extended the standard 30-day response time to [45 or 60 days], as we understand that your capacity to respond may be impacted by the unanticipated COVID-19 pandemic. If you wish to discuss a further extension, please let us know."

Don't hesitate to let us know if you have any questions or concerns.

Thanks,
Kendra

APPENDIX K: NS MUNICIPAL AFFAIRS (NOW DEPARTMENT OF MUNICIPAL AFFAIRS AND HOUSING)

NOVA SCOTIA
Fisheries and Aquaculture

1575 Lake Road
Shelburne, Nova Scotia
Canada B0T 1W0
novascotia.ca

October 7, 2020

Municipality of Antigonish County
ATTN: Glenn Horne, CAO
285 Beech Hill Road
Antigonish, NS
B2G 0B4

Dear Mr. Horne:

Re: Notification of Proposed Marine Aquaculture Application Nos. 1442, 1443, 1444 Antigonish Harbour, Antigonish County

In an effort to keep communities better informed about aquaculture activities in their area, the Province of Nova Scotia is continuing to contact municipalities directly to inform them of proposed applications for new aquaculture sites.

The purpose of this letter is to notify the Municipality of the County of Antigonish of three aquaculture sites located in Antigonish, Harbour, Antigonish County. Please see enclosed information and maps regarding this application.

We do not require your feedback; however, should you have any questions please contact me at 902-875-7430 or by email at Robert.Ceschiutti@novascotia.ca.

Sincerely,

Rob Ceschiutti

Robert Ceschiutti,
Aquaculture Administrator
Nova Scotia Department of Fisheries and Aquaculture

Enclosure: Network Memo and Maps

c. Gordon Smith, Department of Municipal Affairs
Megan Greenwood, Licensing Coordinator
Lew Clancey, Aquaculture Advisor

NOVA SCOTIA

Fisheries and Aquaculture
1575 Lake Road
Shelburne, Nova Scotia
B0T 1W0

novascotia.ca

MEMORANDUM

To: Network Review Agencies

From: Robert Ceschiutti, Aquaculture Administrator
Nova Scotia Department of Fisheries and Aquaculture

CC: Nathaniel Feindel, Manager of Aquaculture Development
Lew Clancey, Aquaculture Advisor
Matthew King, GIS Analyst

Date: October 7, 2020

Re: **Three (3) New Marine Aquaculture Applications (No. 1442, 1443, 1444) – Antigonish Harbour - Aquaculture Network Review**

Attention network agencies, Town Point Consulting Inc. has submitted three (3) new marine aquaculture applications (#1442, #1443 and #1444) for the suspended cultivation of American oyster. The sites are located in Antigonish Harbour, Antigonish County.

Please find attached information relating to the following aquaculture Marine Shellfish applications:

Application No.:	1442, 1443, 1444
Proponent:	Town Point Consulting Inc.
Application Type:	Application for Suspended Cultivation of American oyster
Location:	Antigonish Harbour, Antigonish County

To facilitate the screening process, NSDFA offers the following points of information:

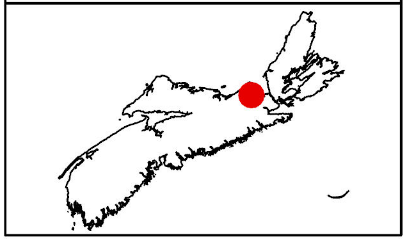
1) Following the review of the application of our Network Partners, this application will be provided to the Aquaculture Review Board (ARB) for final decision;
2) The proponent has made applications (2020-202291, 2020-202293 and 2020-202294) to Transport Canada for an authorization under the *Canadian Navigable Waters Act* for the placement of gear in the water.
3) The complete application package is available online at: https://novascotia.ca/fish/aquaculture/Application-AQ_1442_1443_1444-2020.09.04.pdf
4) Note that although the application package represents the information applicable to all three applications, the NSDFA requests that you provide a separate response for each site, even if the responses are the same. This is due to the method in which the applications are submitted to the ARB.

We request that you review and submit all comments that pertain to this application by **December 7, 2020.**

Note: We require a written (mail/email) response from each of our review agencies in order to process this application.

You may contact Megan Greenwood by phone at 902-875-7443 or by email at Megan.Greenwood@novascotia.ca if you have any questions.

SCHEDULE A

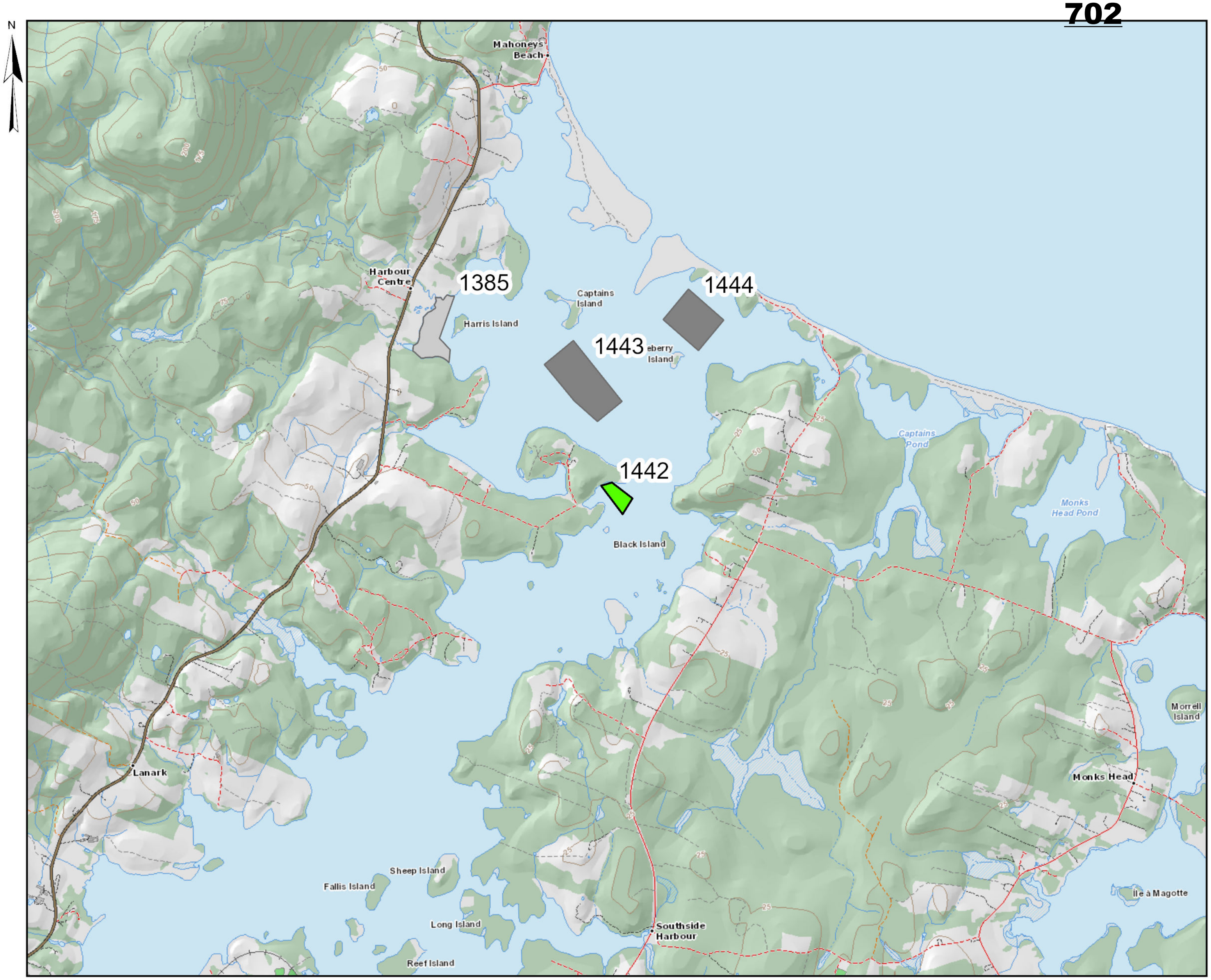


Aquaculture Site

1442

Corner	Latitude	Longitude
1	45° 40' 12.360"	-61° 53' 34.800"
2	45° 40' 16.680"	-61° 53' 30.840"
3	45° 40' 21.104"	-61° 53' 38.809"
4	45° 40' 20.280"	-61° 53' 43.080"
Centre	45° 40' 17.456"	-61° 53' 36.442"

DATUM NAD 83 CSRS UTM Zone 20
The above coordinates are not from a legal survey



License/Lease Holder	County	Waterbody	Hectares	Species Type	Culture Type	Chart
Town Point Consulting Inc.	Antigonish	Antigonish Harbour	3.2	Shellfish	Suspended Culture	4446

🟩 Proposed Application
⬛ Other Proposed Lease
⬜ Other Issued Lease

0 0.5 1 2
Kilometers

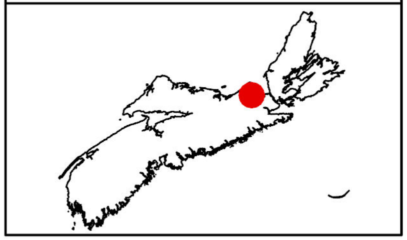
NOVA SCOTIA
Fisheries and Aquaculture

Disclaimer
This map should not be used for navigation or legal purposes. It is intended for general reference use only.

Date: 2020-04-08
Created By: MK

Sources: Esri, HERE, Garmin, Intermap, increment P Corp., GEBCO, USGS, FAO, NPS, NRCAN, GeoBase, IGN, Kadaster NL, Ordnance Survey, Esri Japan, METI, Esri China (Hong Kong), (c) OpenStreetMap contributors, and the GIS User Community, Service Nova Scotia and Internal Services

SCHEDULE A



Aquaculture Site

1442

Corner	Latitude	Longitude
1	45° 40' 12.360"	-61° 53' 34.800"
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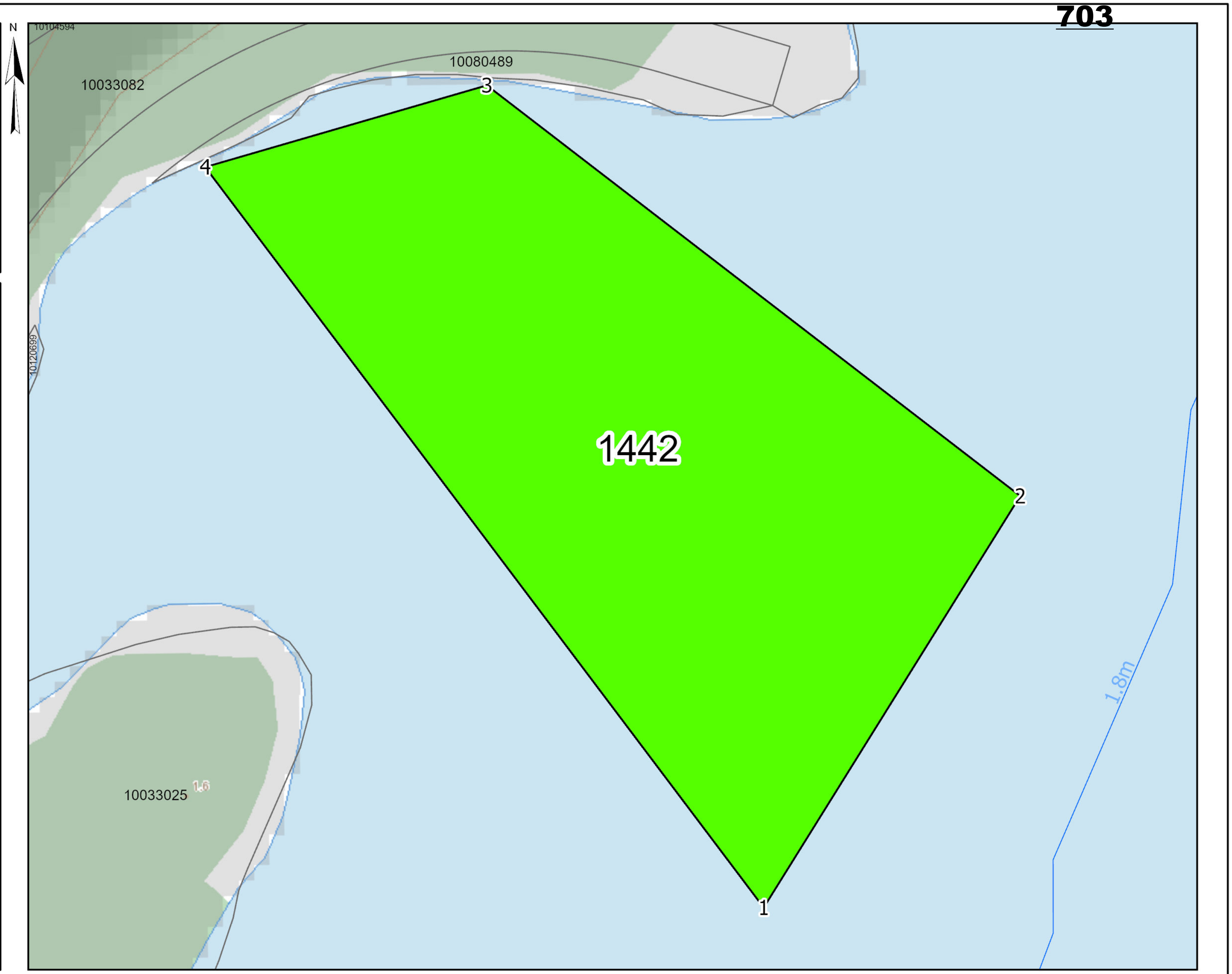

License/Lease Holder	County	Waterbody	Hectares	Species Type	Culture Type	Chart
Town Point Consulting Inc.	Antigonish	Antigonish Harbour	3.2	Shellfish	Suspended Culture	4446

🟩 Proposed Application
⬚ Other Proposed Lease
▨ Other Issued Lease
☐ NS PIDs

0 15 30 60 90
Meters

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Fisheries and Aquaculture

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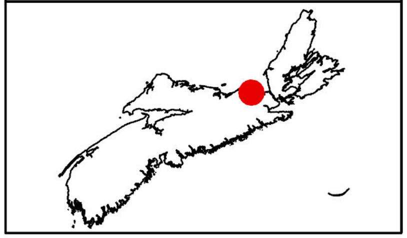
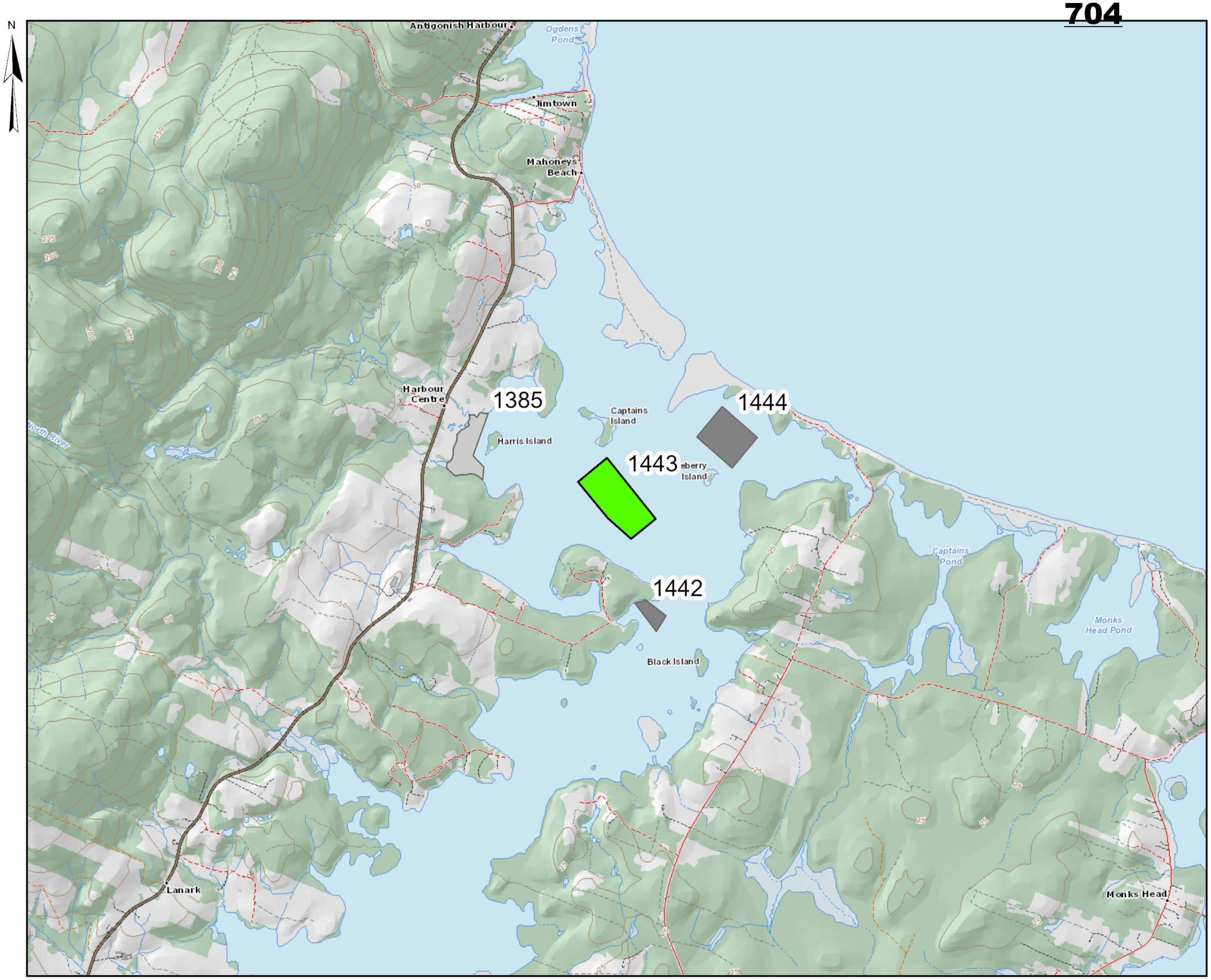


Aquaculture Site

1443

Corner	Latitude	Longitude
1	45° 40' 37.920"	-61° 53' 44.160"
2	45° 40' 43.320"	-61° 53' 34.440"
3	45° 41' 0.240"	-61° 53' 53.160"
4	45° 40' 53.760"	-61° 54' 4.680"
5	45° 40' 43.680"	-61° 53' 53.160"
Centre	45° 40' 48.263"	-61° 53' 49.213"

DATUM NAD 83 CSRS UTM Zone 20
The above coordinates are not from a legal survey



License/Lease Holder	County	Waterbody	Hectares	Species Type	Culture Type	Chart
Town Point Consulting Inc.	Antigonish	Antigonish Harbour	20.21	Shellfish	Suspended Culture	4446

🟩 Proposed Application
■ Other Proposed Lease
▨ Other Issued Lease

0 0.5 1 2
Kilometers

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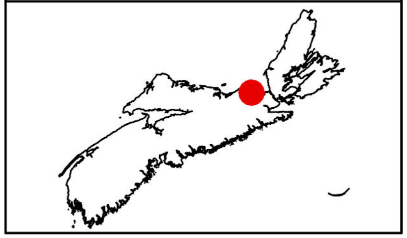

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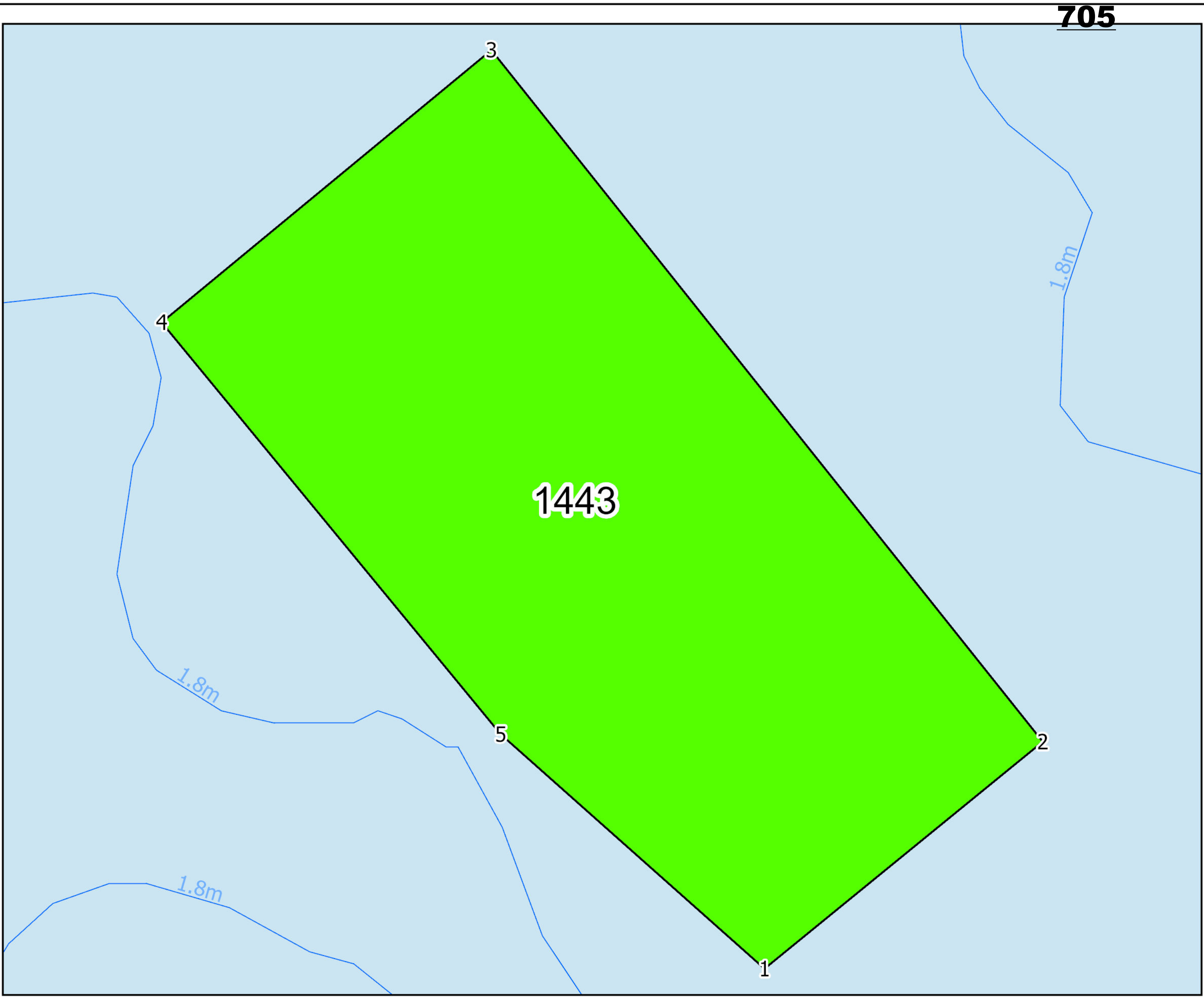

License/Lease Holder	County	Waterbody	Hectares	Species Type	Culture Type	Chart
Town Point Consulting Inc.	Antigonish	Antigonish Harbour	20.21	Shellfish	Suspended Culture	4446

🟩 Proposed Application

▨ Other Issued Lease

⬚ Other Proposed Lease

☐ NS PIDs

0 40 80 160 240
Meters

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Fisheries and Aquaculture

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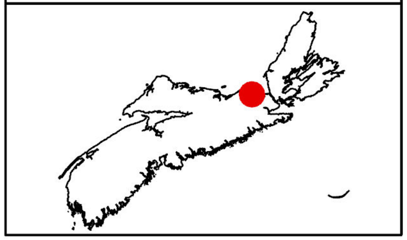
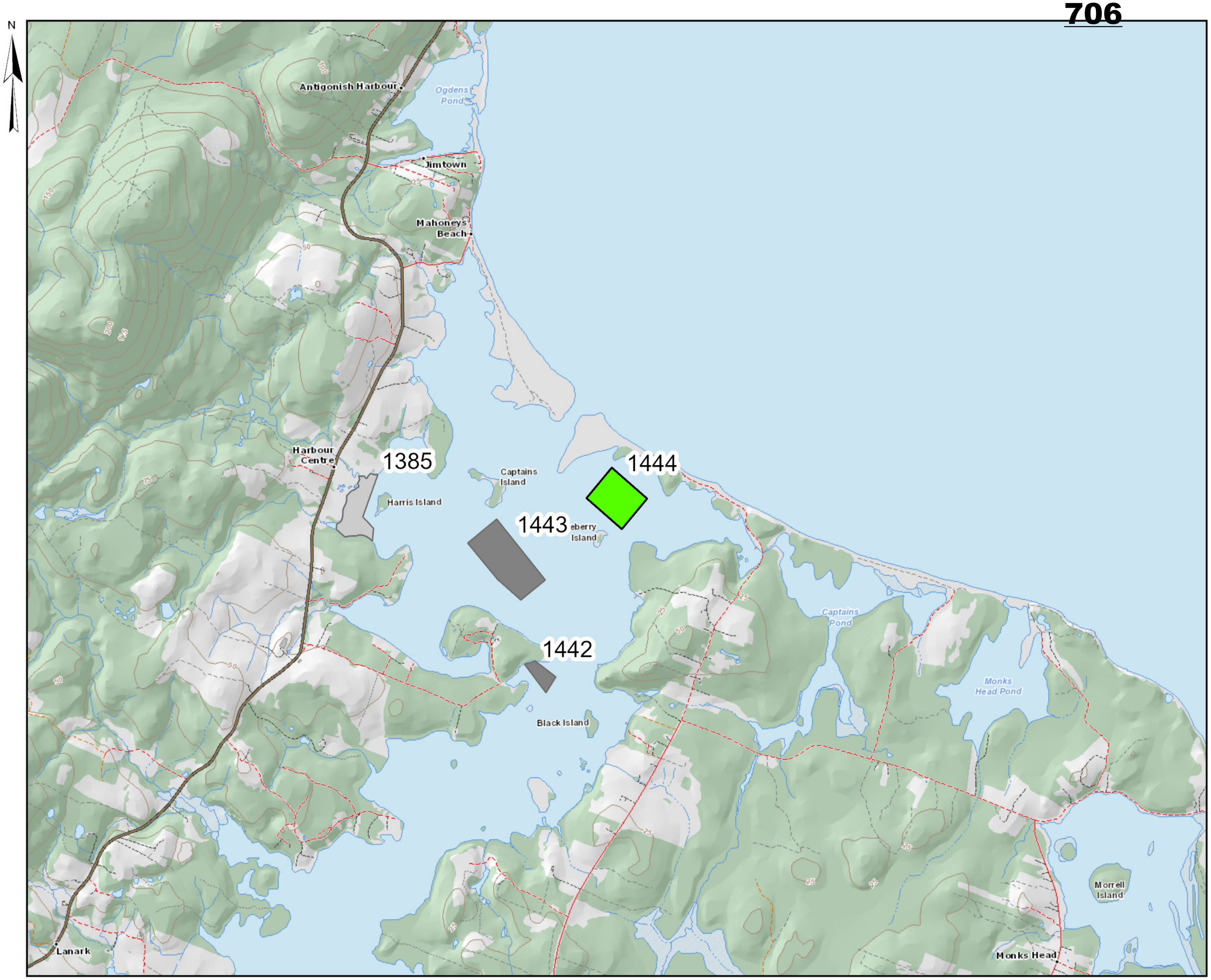


Aquaculture Site

1444

Corner	Latitude	Longitude
1	45° 40' 57.000"	-61° 53' 4.200"
2	45° 41' 5.280"	-61° 52' 54.120"
3	45° 41' 13.920"	-61° 53' 7.800"
4	45° 41' 5.640"	-61° 53' 17.880"
Centre	45° 41' 5.289"	-61° 53' 5.986"

DATUM NAD 83 CSRS UTM Zone 20
The above coordinates are not from a legal survey



License/Lease Holder	County	Waterbody	Hectares	Species Type	Culture Type	Chart
Town Point Consulting Inc.	Antigonish	Antigonish Harbour	13.38	Shellfish	Suspended Culture	4446

🟩 Proposed Application
⬛ Other Proposed Lease
⬜ Other Issued Lease

0 0.5 1 2
Kilometers

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Fisheries and Aquaculture

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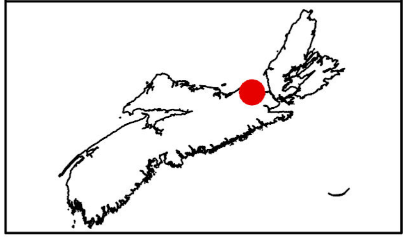
SCHEDULE A



Aquaculture Site

1444

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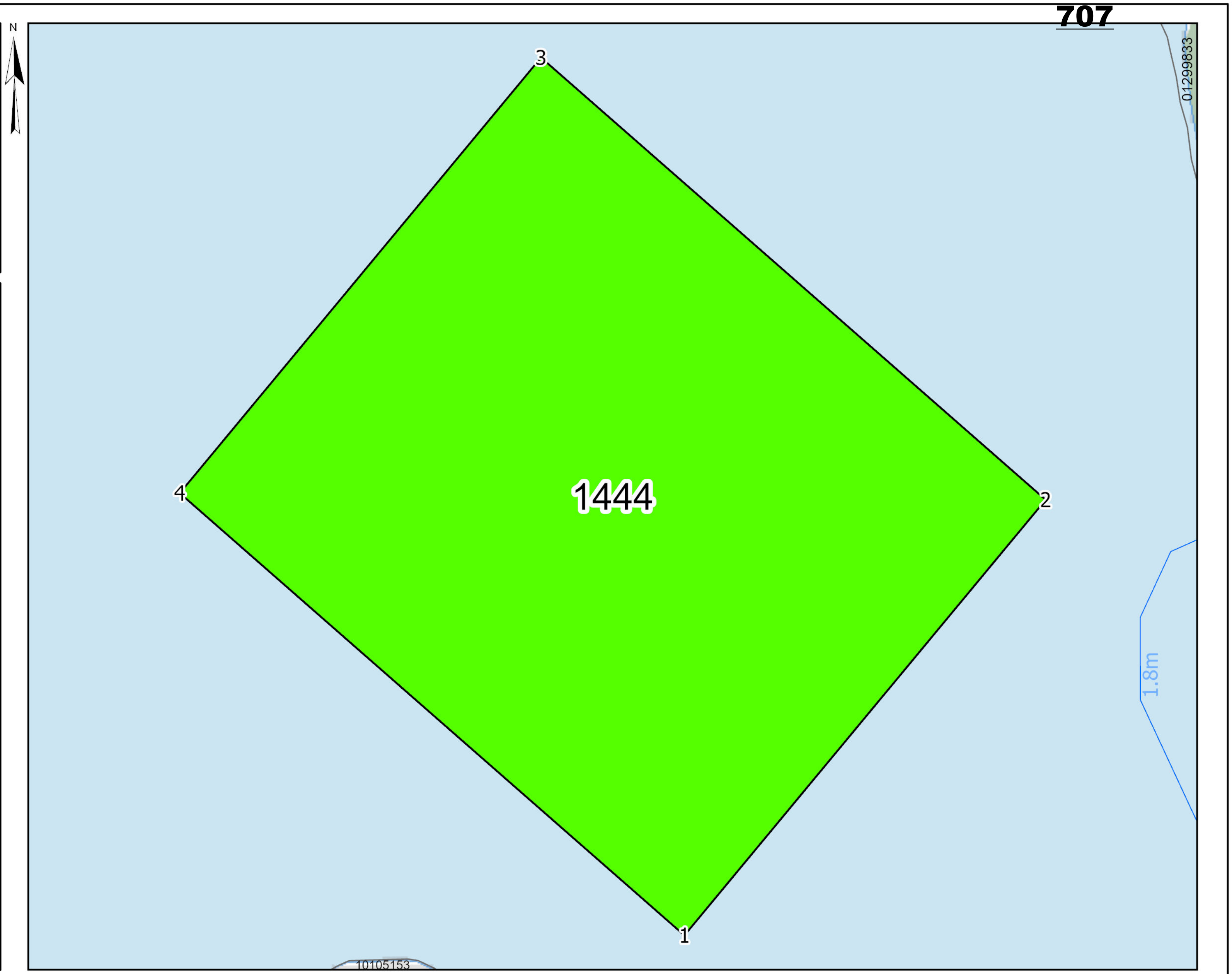

License/Lease Holder	County	Waterbody	Hectares	Species Type	Culture Type	Chart
Town Point Consulting Inc.	Antigonish	Antigonish Harbour	13.38	Shellfish	Suspended Culture	4446

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☐ NS PIDs

0 30 60 120 180
Meters

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Fisheries and Aquaculture

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Date: 2020-04-08
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Sources: Esri, HERE, Garmin, Intermap, increment P Corp., GEBCO, USGS, FAO, NPS, NRCAN, GeoBase, IGN, Kadaster NL, Ordnance Survey, Esri Japan, METI, Esri China (Hong Kong), (c) OpenStreetMap contributors, and the GIS User Community, Service Nova Scotia and Internal Services

APPENDIX L: SAMPLE OF NETWORK MEMO AND NETWORK AGENCY REVIEW OF AN AQUACULTURE APPLICATION

NOVA SCOTIA

Fisheries and Aquaculture
1575 Lake Road
Shelburne, Nova Scotia
B0T 1W0

novascotia.ca

MEMORANDUM

To: Network Review Agencies

From: Robert Ceschiutti, Aquaculture Administrator
Nova Scotia Department of Fisheries and Aquaculture

CC: Nathaniel Feindel, Manager of Aquaculture Development
Lew Clancey, Aquaculture Advisor
Matthew King, GIS Analyst

Date: October 7, 2020

Re: **Three (3) New Marine Aquaculture Applications (No. 1442, 1443, 1444) – Antigonish Harbour - Aquaculture Network Review**

Attention network agencies, Town Point Consulting Inc. has submitted three (3) new marine aquaculture applications (#1442, #1443 and #1444) for the suspended cultivation of American oyster. The sites are located in Antigonish Harbour, Antigonish County.

Please find attached information relating to the following aquaculture Marine Shellfish applications:

Application No.:	1442, 1443, 1444
Proponent:	Town Point Consulting Inc.
Application Type:	Application for Suspended Cultivation of American oyster
Location:	Antigonish Harbour, Antigonish County

To facilitate the screening process, NSDFA offers the following points of information:

1) Following the review of the application of our Network Partners, this application will be provided to the Aquaculture Review Board (ARB) for final decision;
2) The proponent has made applications (2020-202291, 2020-202293 and 2020-202294) to Transport Canada for an authorization under the *Canadian Navigable Waters Act* for the placement of gear in the water.
3) The complete application package is available online at: https://novascotia.ca/fish/aquaculture/Application-AQ_1442_1443_1444-2020.09.04.pdf
4) Note that although the application package represents the information applicable to all three applications, the NSDFA requests that you provide a separate response for each site, even if the responses are the same. This is due to the method in which the applications are submitted to the ARB.

We request that you review and submit all comments that pertain to this application by **December 7, 2020.**

Note: We require a written (mail/email) response from each of our review agencies in order to process this application.

You may contact Megan Greenwood by phone at 902-875-7443 or by email at Megan.Greenwood@novascotia.ca if you have any questions.

Network Agency Review of an Aquaculture Application

Agency	
Division (if applicable)	
Date	
File No.	**1444**
Type of application	**New Marine Shellfish**
Information Provided	

Please provide comments, concerns, recommendations, or requirements on the above stated application for a marine aquaculture licence. Please include the criterion /criteria within your jurisdiction or mandate that your feedback is based upon. Similarly, if additional information is required to make a determination, please include the criterion /criteria within your jurisdiction or mandate that your request is based upon.

☐ No concerns regarding the proposed development
☐ Concerns with development are expressed below
☐ Request modifications to the proposed development (described below)
☐ Required or recommended conditions (described below)
☐ Request additional information (described below)
☐ Request meeting with applicant and NSDFA (described below)
☐ No comments on the application

Comments, concerns, recommendations, and/or required conditions including the criterion /criteria within your jurisdiction or mandate that your feedback is based upon. (Attach comments if preferred, or add additional pages, as required.):

Public Notice and Disclosure

As part of the process for deciding on an application, it may be necessary for the Nova Scotia Department of Fisheries and Aquaculture ("Fisheries and Aquaculture") to disclose the collected network review information to the applicant and other government bodies, including, if applicable, the Nova Scotia Aquaculture Review Board for use at an adjudicative hearing relating to the application in question.

In accordance with departmental policy, which seeks to promote public involvement in the process for deciding on aquaculture applications, Fisheries and Aquaculture will disclose aquaculture application information, including network review information, on the departmental website.

Privacy Statement

The network review information collected as part of an aquaculture application will only be used or disclosed by Fisheries and Aquaculture for the purpose of deciding on the application.

All application information collected is subject to the Freedom of Information and Protection of Privacy Act ("FOIPOP") and will only be used or disclosed in accordance with FOIPOP.