

APPLYING AN ECOSYSTEM-BASED RISK MANAGEMENT APPROACH TO THE
RELATIONSHIP BETWEEN EELGRASS BEDS AND OYSTER AQUACULTURE
AT MULTIPLE SPATIAL SCALES IN EASTERN NEW BRUNSWICK,
ATLANTIC CANADA.

By

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Submitted in partial fulfillment of the requirements for the degree
of
Master of Marine Management

at

Dalhousie University
Halifax, Nova Scotia
January 2014

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In loving memory of
Eleanor & Walter Mitchelle
for seeing me through.

Table of Contents

LIST OF TABLES	V
LIST OF FIGURES	VIII
LIST OF ABBREVIATIONS & GLOSSARY	XI
ABSTRACT	XV
ACKNOWLEDGEMENTS	XVI
CHAPTER 1. INTRODUCTION	1
1.1. COASTAL COMMUNITIES AND INTEGRATED COASTAL ZONE MANAGEMENT	1
1.2. SOUTHERN GULF OF ST LAWRENCE OVERVIEW	2
1.3. CONTEXTUALIZED & SCALE-DEPENDENT RESEARCH	4
1.4. THE MANAGEMENT PROBLEM	5
1.5. RESEARCH PURPOSE, THESIS, AND QUESTIONS	7
CHAPTER 2. METHODS	8
2.1. RESEARCH STRATEGY / METHODS SUMMARY	8
2.2. LITERATURE REVIEW	10
2.3. DATA COLLECTION	11
2.3.1. <i>Near-Field Scale Data</i>	11
2.3.2. <i>Zone of Influence Scale Data</i>	12
2.3.3. <i>Bay, Estuary, and Watershed Scale Data</i>	13
2.4. DATA ANALYSES	15
2.4.2. <i>Statistical Tests</i>	17
2.5. RISK ANALYSES	21
2.6. MANAGEMENT OPTIONS AND RECOMMENDATIONS	22
CHAPTER 3. LITERATURE REVIEW	22
3.1. THE SOUTHERN GULF OF ST LAWRENCE AND EASTERN NB: THE SOCIAL-ECOLOGICAL UNIT	22
3.2. VALUABLE ECOSYSTEM COMPONENTS AND SERVICES	24
3.2.1. <i>Eelgrass</i>	24
3.2.2. <i>American Oysters</i>	24
3.3. LEGISLATION, POLICIES, MANAGEMENT OUTCOMES, AND GOVERNANCE STRUCTURES	27
3.4. SIGNIFICANT ECOSYSTEM COMPONENT SUSCEPTIBILITIES	30
3.5. CAUSE-EFFECT PATHWAYS	31
3.5.1. <i>Near-field Scale SBOA Effects</i>	34
3.5.2. <i>Zone of Influence Scale SBOA Effects</i>	35
3.5.3. <i>Bay Scale SBOA Effects</i>	36
3.5.4. <i>Estuary Scale SBOA Effects</i>	37
CHAPTER 4. RISK ASSESSMENT RESULTS	40
4.1. NEAR-FIELD SCALE RISK ASSESSMENT	40
4.2. ZONE OF INFLUENCE SCALE RISK ASSESSMENT	43
4.5. SUMMARY OF RISK ASSESSMENT RESULTS	52
CHAPTER 5. DISCUSSION	55
5.1. BAY- AND ESTUARY-SCALE EFFECTS	57
5.2. RISK ASSESSMENTS	59
5.3. CAVEATS AND LIMITATIONS	60
5.4. MANAGEMENT RECOMMENDATIONS	63

CHAPTER 6. CONCLUSION	68
CHAPTER 7. REFERENCES	71
CHAPTER 8. APPENDIX A.....	80
8.1. METHODS	80
8.1.1. <i>Bay and estuary scale data</i>	80
8.1.2. <i>Watershed Data</i>	81
8.1.3. <i>Data and Statistical Analyses</i>	82
8.1.4. <i>Risk Assessment Criteria and Examples</i>	87
8.3. PATHWAYS-OF-EFFECTS	90
8.4. RESULTS.....	90
8.4.1. <i>Manipulated SDI Numerical Model</i>	90
8.4.2. <i>Bay-Scale Statistical Analyses</i>	93
8.4.3. <i>Estuary-Scale Statistical Analyses</i>	104
CHAPTER 9. APPENDIX B.....	119
9.1. R-STUDIO CODE FOR BAY-SCALE STATISTICS.....	119
9.1.1. <i>Loading Data</i>	119
9.1.2. <i>Multiple Regression Analysis</i>	119
9.1.3. <i>Eelgrass Predictions</i>	119
9.1.4. <i>Principal Components Analysis</i>	119
9.1.5. <i>Sensitivity Analysis</i>	120
9.2. R-STUDIO CODE FOR ESTUARY-SCALE STATISTICS	122
9.2.1. <i>Loading Data</i>	122
9.2.2. <i>Multiple Regression Analysis</i>	122
9.2.3. <i>Principal Components Analysis</i>	123
9.3. R-STUDIO CODE FOR WATERSHED-SCALE STATISTICS.....	125
9.3.2. <i>Principal Components Analysis</i>	126

List of Tables

- Table 1. List of the eastern New Brunswick estuaries (n=8) and bays (n=14) used for statistical analyses9.3.2. Principal Components Analysis 126
- Table 2. List of dependent and independent variables used for statistical analyses at the bay (n=11; n=6 raw data and n=5 interaction terms), estuary (n=6), and watershed (n=6) scales. Abbreviations and expressed units for the variables are in parentheses 19
- Table 3. Area of eelgrass cover (ha) per bay (n=14) from EC/DFO data and the GLM model predictions. The % difference shows the difference of the eelgrass cover predictions relative to the EC/DFO estimates9.3.2. Principal Components Analysis 19
- Table 4. Risk analysis of the near-field effects of SBOA onto underlying eelgrass meadows. The likelihood has been ranked as a 4 (likely) dependent on the fact that 76-95% of case studies demonstrated significant changes between/under leases compared to reference sites. The overall impacts and consequences for near-field effects have been ranked as a 3 (medium; rounded from 2.85) upon averaging the individual rankings for the ecological, socio-economic, operations, and strategic consequences 42
- Table 5. Risk analysis of the zone of influence (ZOI) effects of SBOA onto surrounding eelgrass meadows. The likelihood has been ranked as a 4 (likely) given that it is assumed that 76-95% of case studies would demonstrate similar ZOI effects. The overall impacts and consequences for ZOI-scale effects have been ranked as a 1 (negligible; rounded from 1.25) upon averaging the individual rankings for the ecological, socio-economic, operations, and strategic consequences 44
- Table 6. Risk analysis of the bay-scale effects of SBOA onto eelgrass cover. The statistical analyses have revealed a significant probability between SBOA characteristics and eelgrass cover ($P=>2.16e-16$), and so the likelihood of bay-scale SBOA effects was ranked as a 5 (almost certain; >95% likelihood). The overall impacts and consequences for bay-scale effects have been ranked as a 2 (low; rounded from 2.38) upon averaging the individual rankings for the ecological, socio-economic, operations, and strategic consequences 47
- Table 7. Risk analysis of the bay-scale effects of SBOA onto eelgrass cover. The statistical analyses have revealed a significant probability between SBOA characteristics and eelgrass cover ($P=>2.16e-16$), although without better integrating the effects of watershed processes the estuary-scale likelihood of risk was conservatively ranked as a 4 (likely; 76-95% likelihood). The overall impacts and consequences for near-field effects have been ranked as a 2 (low; rounded from 1.86) upon averaging the individual rankings for the ecological, socio-economic, operations, and strategic consequences 51
- Table 8. A summary of the Environment Canada and Fisheries and Oceans Canada eelgrass data collection methods for eastern NB estuaries (n=8) from 2007-2009.

Classification categories: GQ = Good Quality, MQ = Medium Quality, PQ = Eelgrass absent/Poor Quality; DE = Dense Eelgrass, ME = Moderate Eelgrass, TE = Thin Eelgrass, EE = Exposed Eelgrass; EP = Eelgrass Presence. Table adapted from AGRG, 2012; St-Simon was added. Adapted from the AGRG (2012) report with additions..... 81

Table 9. List of Natural Resources Canada (NRC) abbreviations and descriptions of primary land-uses of non-forested areas commonly found within each watershed in eastern New Brunswick 81

Table 10. List of raw dependent and independent data used for the bay-scale statistical analyses encompassing the n=14 bays along eastern NB. ASL = Active suspended lease area; TSL = total suspended lease area; TAL = total aquaculture lease area 83

Table 11. List of raw dependent and independent data used for estuary-scale statistical analyses encompassing the n=8 estuaries along eastern NB. Table A) n=6 estuary data and B) watershed data. ASL = Active suspended lease area; TSL = total suspended lease area; TAL = total aquaculture lease area. AGR = agriculture area; IND = industrial area; INF = infrastructure area; SET = settlement area 83

Table 12. The calculated raw and normalized means (%) of each tested SBOA co-variable used for the sensitivity analyses. Bay-scale analyses examined n=5 co-variables for a total of n=10 sensitivity tests; estuary-scale analyses examined n=4 co-variables as it had omitted gear area means, thus for a total of n=8 sensitivity tests 86

Table 13. Summary of the hypothesized ecological implications of the eelgrass cover responses to a range of manipulated SBOA variables 87

Table 14. Risk management likelihood criteria for the n=5 likelihood categories. Adapted from Cormier et al., 2013..... 87

Table 15. Risk management criteria for each of n=5 risk categories that describe the degree of the ecological impacts, socio-economic consequences, and operational and strategic repercussions. Adapted from Cormier et al., 2013 88

Table 16. An example of the risk analysis for coastal eutrophication case study based upon the risk likelihood and assessment criteria (see Table 14 and 15, above). Adapted by Cormier et al., 2013; see for more examples of risk assessments 89

Table 17. Summary table of the linear and multiple step-wise regression results at the bay-scale using the n=11 bay-scale raw and interacting SBOA co-variables. Significant results are indicated with bold text..... 94

Table 18. The percentage of variance explained by the n = 6 dimensions from the bay-scale Principal Components Analysis..... 96

Table 19. The sensitivity analysis results at the bay-scale, representing each of the n = 5 manipulated SBOA values, expressed in both mean area (ha) and mean percent predicted eelgrass cover per bay (%) with n = 14 bays per analysis. Results are compared to mean EC/DFO eelgrass cover data to create the mean percent difference in eelgrass cover (% difference) in response to an increase or decrease in the SBOA values..... 98

Table 20. Summary table of the linear and multiple step-wise regression (MRA) results at the estuary scale using the n=6 estuary and n=6 watershed-scale SBOA co-variables. Significant results are indicated with bold text. Gear area was dropped from the MRA; the MRA was not preformed using the watershed co-variables due to high correlations..... 104

Table 21. The percentage of variance explained by each of the n = 6 dimensions resulting from the estuary-scale Principal Components Analysis 106

Table 22. The percentage of variance explained by each of the n = 6 dimensions resulting from the watershed-scale Principal Components analysis..... 108

Table 23. The mean SA results for each of the n = 10 manipulated SBOA values, expressed in both area (ha) and percent predicted eelgrass cover per bay (%) with n = 14 bays per analysis. Results are compared to mean EC/DFO eelgrass cover data to create the mean percent difference in eelgrass cover (% difference), providing information as to the mean growth and/or decline in response to an increase or decrease in the SBOA values 110

List of Figures

- Figure 1. Map of the eastern New Brunswick coastline. The n=8 estuaries outlined in blue were examined for the purpose of this study; estuaries comprised of >1 bay were enlarged to illustrate subdivided bay boundaries. A total of n=14 bays were examined. Adapted from DFO, 2011 9
- Figure 2. Map of the Richiboucto estuary illustrating the four spatial scales examined therein. The blue, yellow, and green contours describe the approximate boundaries of each of the three bays – Aldouane, Richiboucto Harbour, and Bedec, respectively – while the purple circle depicts the one of the potential zone of influence scales present within Bedec. The white outlines are the suspended bag oyster aquaculture lease areas and represent the near-field scale. 10
- Figure 3. Map of Atlantic Canada with focus on the eastern New Brunswick (NB) coastline along the southern Gulf of St Lawrence (sGSL). Adapted from Skinner et al., 2013 23
- Figure 4. Number of annual oyster aquaculture bag equivalents reported from 1996-2009 along NB’s eastern shoreline. Adapted from Comeau, 2013 26
- Figure 5. A legislative profile of relevant Canadian federal and provincial acts and regulations that pertain to coastal and marine spatial planning and management along the New Brunswick coastline of the southern Gulf of St Lawrence. Adapted from Cormier, 2010 29
- Figure 6. Risk analysis heat map for the near-field effects of SBOA onto eelgrass. Given that the risk analysis results have demonstrated a “likely” likelihood, and that the overall impacts and consequences averaged to be “medium”, the near-field eelgrass-SBOA relationship exudes “not tolerable” conditions and thus demands ecosystem-based risk management and treatment options 43
- Figure 7. Risk Analysis Heat Map for the ZOI effects of SBOA onto eelgrass. Given that the risk analysis results have demonstrated a “likely” likelihood, and that the overall impacts and consequences averaged to be “negligible”, the ZOI-scale eelgrass-SBOA relationship exudes “extremely tolerable” conditions and does not demand immediate ecosystem-based risk management and treatment options 45
- Figure 8. Risk Analysis Heat Map for the bay-scale effects of SBOA onto eelgrass. Given that the risk analysis results have demonstrated an “almost certain” likelihood, and that the overall impacts and consequences averaged to be “low”, the near-field eelgrass-SBOA relationship exudes “tolerable” conditions and does not demand immediate ecosystem-based risk management and treatment options 48
- Figure 9. Risk Analysis Heat Map for the estuary-scale effects of SBOA onto eelgrass. Given that the risk analysis results have demonstrated a “likely” likelihood, and that the overall impacts and consequences averaged to be “low”, the estuary-scale eelgrass-SBOA

relationship exudes “tolerable” conditions and does not demand immediate ecosystem-based risk management and treatment options 51

Figure 10. Map of the n=38 watersheds that influence the southern Gulf of St Lawrence. Watersheds 1-17 are within the provincial boundaries of New Brunswick; therein, watersheds 5-6, 8-9, and 12-14 influence the n=8 estuaries examined for the purpose of the current research project..... 82

Figure 11. An example of the risk heat map indicating non-tolerable conditions in response to the coastal eutrophication risk assessment case study (see Fig. 11, above). Red blocks are indicative of extremely not tolerable conditions; orange is not tolerable; yellow is tolerable; and, green is extremely tolerable. See the IOM Risk Management Handbook (Cormier et al., 2013) for more examples of heat maps 89

Figure 12. A conceptual model of the various cause-effect pathways for HADD onto eelgrass. The yellow nodes represent the cumulative disturbances and pressures caused by the multiple competing human activities and uses; the orange nodes are the stressors and impacts; the blue node is the response of HADD onto eelgrass; the green nodes are the socio-economic sectors that experience repercussions due to the loss of eelgrass..... 90

Figure 13. Digital areal images illustrating the seston depletion index of the Richiboucto estuary in response to different SBOA conditions produced by the numerical model (Guyondet et al., 2013). A) with SBOA present in Aldouane only; B) with SBOA present in Bedec only; C) with SBOA leases throughout the Richiboucto estuary 92

Figure 14. The water renewal time for areas of the Richiboucto estuary measured in days. Adapted from Guyondet et al., 2013..... 93

Figure 15. A residuals against fitted values (RAFV) plot from the bay-scale multiple regression analysis. Data was found distributed in a horizontal band around the residuals = 0 line, illustrating that the variances of the error terms were equal as well as the data having a relatively linear relationship. Bays 2 (St Simon Inlet), 3 (St Simon North), and 11 (Richiboucto Harbor) are outliers 95

Figure 16. The Q-Q plots resulting from the bay-scale multiple regression analysis. The data was distributed along the residuals = 0 line ($y = x$), indicating that the eelgrass data and MRA model were significantly similar to one another. The deviation from the line at both tails of the plot indicate the presence of outliers at both extremes of the data; bays 2 (St Simon Inlet), 7 (Tracadie South), and 8 (Tabusintac) are outliers 95

Figure 17. A variable factor map produced from a Principal Components analysis on the standardized bay-scale SBOA variables. The order of the co-variables’ labels is in order of the eigen vectors, beginning with zASL and moving clock-wise. Dimension 1 explains 42.8% of the overall variance; dimension 2 explains 36.0% of the variance 96

- Figure 18. Histograms of the percent eelgrass cover per bay area (%) for each of the $n = 14$ bays as the sensitivity analysis responds to the manipulated SBOA co-variables. The ground-truth EC/DFO eelgrass cover is plotted alongside the predicted eelgrass cover results from the $n=10$ sensitivity analyses that examined the effects of minimum and maximum values for each of the $n=5$ SBOA co-variables: (A) ASL, (B) TSL, (C) TAL, (D) bag counts, (E) gear area 101
- Figure 19. A plot of residuals against fitted values resulting from the estuary-scale multiple regression analysis. Data distribution was shaped as a horizontal band above the residuals = 0 line, illustrating a relatively linear relationship within the data although the variances of the errors terms were not found to be equal. Bay 4 (Tabusintac) is an outlier..... 105
- Figure 20. The Q-Q plots resulting from the estuary-scale multiple regression analysis. The data was found distributed along the residuals = 0 line ($y = x$), indicating that the eelgrass data and MRA model were significantly similar to one another. The deviation from the line at both tails of the plot indicate the presence of outliers at both extremes of the data; bays 4 (Tabusintac) and 8 (Cocagne) are outliers..... 105
- Figure 21. A variable factor map (VFM) produced from a Principal Components analysis on the standardized estuary-scale SBOA variables. The order of the co-variables' labels is in order of the eigen vectors, beginning with z-estuary area and moving clockwise. Dimension 1 explains 61.7% of the overall variance; dimension 2 explains 19.6% of the variance 107
- Figure 22. A variable factor map (VFM) produced from a Principal Components analysis on the standardized watershed-scale SBOA variables. The order of the co-variables' labels is in order of the eigen vectors, beginning with z-watershed area and moving clockwise. Dimension 1 explains 62.2% of the overall variance; dimension 2 explains 20.7% of the variance 109
- Figure 23. Histograms of the percent eelgrass cover per estuary area (%) for each of the $n = 8$ estuaries as the sensitivity analysis responds to the manipulated SBOA co-variables. The ground-truth EC/DFO eelgrass cover is plotted alongside the predicted eelgrass cover results from the $n=8$ sensitivity analyses that examined the effects of minimum and maximum values for each of the $n=4$ SBOA co-variables: (A) ASL, (B) TSL, (C) TAL, (D) bag counts..... 112

List of Abbreviations & Glossary

AIC – Akaike Information Criterion

A statistical tool to assess the quality of a model, and therefore represents a tool for model selection.

ASL – Active Suspended Bag Oyster Aquaculture Lease Area

The area that is actively used to culture oysters; distinct from the area leased for oyster culture but is left vacant.

Bay

A partially enclosed body of water and often a subset of a larger estuary (see definition below), with distinct hydrodynamic and oceanographic processes and conditions from other bays belonging to the same estuary.

CEAA – Canadian Environmental Assessment Act

CRA – Commercial, Recreational, and Aboriginal Fisheries

CT – Clearance Time

“The number of days required for the dominant bivalve stock(s) (wild and cultured) to clear the volume of the bay or regional water body (i.e., sites with no clear boundaries).” (ASC, 2013, p. 16)

DFO – Department of Fisheries and Oceans Canada

EBSA – Ecologically and Biologically Significant Areas

“Identifying Ecologically and Biologically Significant Areas is not a general strategy for protecting all habitats and marine communities that have some ecological significance. Rather, it is a tool for calling attention to an area that has particularly high Ecological or Biological Significance, to facilitate provision of a greater-than- usual degree of risk aversion in management of activities in such areas.” (DFO, 2004, p. 1)

EBRM – Ecosystem-Based Risk Management

According to Cormier and colleagues (2013, p. 15), the five core elements are:

- 1. Recognizing connections among marine, coastal, and terrestrial systems, as well as between ecosystems and human societies;*
- 2. Using an ecosystem services perspective, where ecosystems are valued for the basic goods they generate, as well as for the important services they provide;*
- 3. Addressing the cumulative impacts of various activities affecting an ecosystem;*
- 4. Managing and balancing multiple and sometimes conflicting objectives that are related to different benefits and ecosystem services; and,*
- 5. Embracing change, learning from experience, and adapting policies throughout the management process.*

EC – Environment Canada

ECC – Ecological Carrying Capacity (relevant to oyster aquaculture)

“Ecological carrying capacity has been defined as the stocking or farm density above which unacceptable ecological impacts begin to manifest (ASC, 2013, p 16)

ESS – Ecologically Significant Species

“The identification of Ecologically Significant Species (ESS) is a tool for calling attention to a species or community property that has particularly high ecological significance. DFO held a workshop in September 2006 to review and develop criteria to differentiate species or properties which are “particularly important” or “significant” with regards to specific ecosystem structure and function”. (DFO, 2009, p 1)

Estuary

“A partially enclosed body of water in the lower reaches of a river, which is freely connected with the sea and which receives fresh water supplies from upland drainage areas.” (Transport Canada, 2007, p 62)

ICZM - Integrated Coastal Zone Management

“Canada’s Oceans Act promotes an Ecosystem Approach to the integrated management of human activities. Integrated management plans must include objectives intended to protect the ecosystem. As such, enhanced protection should be provided to species and community properties that are particularly significant to maintaining ecosystem structure and function.” (DFO, 2009, p.1)

GSL – Gulf of St Lawrence

GLM – General Linear Model

HADD – Harmful Alteration, Disruption and Destruction

“Section 35 of the Fisheries Act (1986), which prohibits the [HADD] of fish habitat, provides the Minister with the power to authorize terms and conditions which would allow projects to proceed in compliance with the Act... A HADD is whereby the biophysical attributes of fish habitat are modified such that the habitat is rendered less suitable for fish production.” (DFO, 2002, p. 1-2)

LMA – Line of Moving Averages

A tool to help visualize the significance of the mean as it changes across observations in a plot.

MOU – Memorandum of Understanding

MRA – Multiple Regression Analysis

NB – New Brunswick

NB AAF – NB Department of Agriculture, Aquaculture, and Fisheries

NRC – Natural Resources Canada

PCA – Principal Component Analysis

PPT – Primary Productivity Time

“The number of days required for the replacement of the standing stock of phytoplankton in the bay (i.e., time-scale of phytoplankton population growth).” (ASC, 2013, p 16)

RAFV – Residuals Against Fitted Values (plot)

RCSR – Replacement Class Screening Report

“Transport Canada (2007) developed a Replacement Class Screening Report (RCSR) for “water column oyster aquaculture in New Brunswick” that described mitigation measures that if applied would eliminate any significant residual effects of the activities on the functioning of eelgrass as fish habitat. The three residual environmental effects which required mitigation included physical removal of the plant, damage to the plant by infrastructure and operations, and shading of the plant by the infrastructure.” (DFO, 2011, p 1) An update to the RCSR was expected in 2012, but the program was terminated beforehand.

RT – Retention Time

“The number of days for tides to flush a volume of water equal to the volume of the bay or water body.” (ASC, 2013, p 16)

SA – Sensitivity Analysis

SBOA – Suspended Bag Oyster Aquaculture

A form of aquaculture conducted in the water column or at the surface, where the structures are anchored but float or move with the tides (Transport Canada, 2007, p 63)

SES – Social-Ecological System

Defined by the German Society for Human Ecology (DFH) has developed a working definition for a social-ecological system (SES) that purports, “a [SES] consists of a bio-geo-physical [ecosystem] unit and its associated social actors and institutions. [SES] are complex and adaptive and delimited by spatial or functional boundaries surrounding particular ecosystems and their problem context” (as cited by Jahn, 2005, 2).

sGSL – southern Gulf of St Lawrence

TAL – Total Oyster Aquaculture Lease Area

The geographical area of coastal zones leased for the use of oyster aquaculture, including both suspended bag and bottom culture techniques.

TSL – Total Suspended Bag Oyster Aquaculture Lease Area

The geographical area of coastal zone leased for the specific use of suspended bag oyster aquaculture, including both the active and non-active leased area.

VEC – Valued Environmental and Socio-Economic Component

“The environmental element of an ecosystem that is identified as having scientific, social, cultural, economic, historical, archaeological or aesthetic importance.” (DF0, 2013)

VFM – Variable Factor Map

Watershed

“A geographic concept designating a territory whose land is drained by any one body of water, such as a bay or a river, and which includes groundwater, surface water and wetlands.” (Transport Canada, 2007, p 63)

ZOI – Zone of Influence

A geographical area wherein the aquaculture activities and processes significantly affect the biological, chemical, physical, and ecological aspects of the coastal zone and its values.

Vance, A. 2014. Applying an ecosystem-based risk management approach to the relationship between eelgrass beds and oyster aquaculture at multiple spatial scales in eastern New Brunswick, Atlantic Canada [graduate project]. Halifax, NS: Dalhousie University.

ABSTRACT

Both eelgrass beds (*Zostera marina*) and the American Oyster (*Crassostrea virginica*) are indigenous to Atlantic Canadian coastal waters and are equally recognized as ecological engineers. However, recent eelgrass cover declines and simultaneous increases in the suspended-bag oyster aquaculture (SBOA) industry in coastal eastern New Brunswick (NB) has potentially disrupted various coastal ecosystem services and functions. This research examined the ecological and biophysical relationship between eelgrass cover and local SBOA effects using an ecosystem-based risk management (EBRM) approach at four distinct spatial scales; the near-field, zone of influence, bay, and estuary scales. Using the available literature, regional data, and multi-variate statistical analyses, the relative impact to eelgrass cover at each examined spatial scale was assessed for the risk of ecological, socio-economic, strategic, and operational consequences of the SBOA industry. The results suggest that the eelgrass-SBOA relationship is both positive and negative depending on the scale observed, and that significant trade-offs exist both within and among each spatial scale. This research has provided a preliminary examination of the ecological status and response to SBOA, and has recommended a number of integrated coastal zone management (ICZM) measures, including: 1) using standardized data collection methodologies, 2) integrating stakeholders and their local knowledge into decision-making, 3) implementing best practices, such as the use of less intensive gears and better spacing, and 4) mandating contextualized bay-scale ICZM plans. These recommendations have been offered to ensure the long-term sustainable development of the SBOA industry and the health of local eelgrass ecosystems throughout eastern NB and Atlantic Canada.

Keywords: Atlantic Canada; ecosystem-based management; risk analysis; carrying capacity; eelgrass (*Zostera marina*); oyster aquaculture; sustainable industry development

Acknowledgements

There are several people whose contributions to this project made it possible. First, those at DFO and in particular the work of Monique Niles and Andrea Locke, who had proposed the initial version of this research and compiled the bay, estuary, and watershed-data for analyses. I would like to especially thank Dr Heike Lotze; without her support, professional insight, and funding, this project would have certainly derailed long ago. I am grateful for the inequivalent statistical mentorship of Dr Stu Carson and especially Joey Hartling, who had spent months of long days to ensure that I not only had completed all of the analyses, but understood them as well. I would also like to thank Dr Marc Skinner and Thomas Guyondet, who had both graciously granted me early access to their research so that I could incorporate their results into my own; without these results, this project would have been incomplete. The inspiring and holistic perspectives of Marc Ouellette and Roland Cormier had guided me to adopt the EBRM framework used to structure this project. I am also thankful for the efforts made by Guy Robichaud and John Brazner, who helped guide me through eelgrass classifications and various watershed impacts, respectively. Last but certainly not least is my supportive network of loving family and friends, with the unconditional encouragement of my mother that held it all together. And to my dearest friends Eleanor and Walter Mitchell, who have made my education possible and to whom I proudly thank for this accomplishment. May you both rest in peace.

1. Introduction

1.1. Coastal Communities and Integrated Coastal Zone Management

Simultaneously recognized as a valuable yet vulnerable social-ecological systems (SES), coastal communities around the world are amongst the most productive systems (Costanza et al., 1997) and offer an abundance of critical services and functions that in turn benefit and support a variety of socio-economic activities (IPCC, 1996). However, recent growth and development of the coastal zone has created disproportionately large pressures onto this SES, causing a suite of effects and impacts onto both the social and ecological sub-systems (Hinrichsen, 1998; IPCC, 1996, UNEP, 2006). One estimate suggests that approximately half of the global population now lives within 150 km of the coastline, which encompasses less than 10% of global land cover (Halpern et al., 2008). There is high confidence that as a result of increased coastal development and habitation, the adaptive capacity and resilience of the SES has decreased and has rendered it less able to cope with natural climate variability, climate change, and further resource use and human activity (IPCC, 1996; UNEP, 2006). Moreover, the increasingly multiplicative and overlapping use and activity of the coastal zone has inevitably created coastal conflicts, which are preventing the sustainable use and development of the SES. These effects are in turn increasing the hazard potential for coastal social communities, their investments and infrastructure (IPCC, 1996). Ultimately, the SES and its social and ecological sub-systems are inextricably linked (Folke et al., 2010) and the cumulative effects of the multiple uses are reflected in the relatively high impact of heavily populated coastal zones (Halpern et al., 2008).

The multiple competing coastal resources' uses and activities have created a complex problem for coastal managers. The ability to assess the interdependent effects onto the SES becomes increasingly difficult because the impacts may be slow to emerge or emerge at some distance from where the ecosystem was impacted, and because the social consequences differ for different sets of stakeholders (Millennium Ecosystem Assessment, 2005). Therefore, impacts to one sub-system and its respective management solution may catalyze another, potentially more complex response in another sub-system (MEA, 2005; Kearney and Berkes, 2007). However, due to

reductionist perspectives and the institutional concentration of disciplines, managers and decision-makers have largely managed these sectoral issues in isolation of the larger complex SES (Vallega, 2001; Kearney and Berkes, 2007; Pohl, 2011). As a result, social and ecological sub-systems have been managed separately from one another, despite widespread recognition of the interdependencies of the SES (MEA, 2005; UNEP, 2006). The traditional sectoral coastal management approaches have thus become less effective over time given the increasingly vulnerable, complex, and overlapping nature of the coastal zone and its SES impacts.

Ecosystem-based risk management (EBRM) approaches (see Appendix 10.1. for definition) have been developed in response to international recognition of the need to balance the ecological integrity and sustainable development of the socio-economic activities. For estuarine, coastal, and marine areas and their SES, a more specific guiding framework has been developed to address their particular management challenges: integrated coastal zone management (ICZM). ICZM frameworks are internationally supported (e.g., Agenda 21) given that they facilitate sustainable coastal development by interconnecting various levels of government and governance, economic and industrial sectors, academic disciplines, stakeholders, resource users, and generations (UNEP, 1995; GESAMP, 1996; World Bank, 1996; Costanza et al., 1997; Crooks and Turner, 1999; Wiedemeyer, 2010). If contextually developed, an ICZM framework can strategically mitigate socio-economic pressures and effects onto the coastal ecosystem while sustainably maximize the SES benefits of ecological services and functions (Wiedemeyer, 2010). As human populations continue to concentrate themselves in coastal zones and catalyze development, it will become increasingly important approach growth with an EBRM perspective and implement appropriate ICZM measures.

1.2. Southern Gulf of St Lawrence Overview

Atlantic Canadian coastal waters encompass a myriad of rich coastal and marine ecosystems; in particular, the Gulf of St Lawrence (GSL) and specifically the southwestern region of the GSL (sGSL) have been federally and formally recognized as having a unique and complex SES (DFO, 2005; DFO, 2007). As such, the Department of Fisheries and Oceans Canada (DFO) is legislatively mandated to implement and

promote ICZM throughout the sGSL to ensure the sustainable development of these coastal and marine ecosystems and the socio-economic activities that depend upon them (DFO, 2005; DFO, 2007). Therein, DFO recognized eelgrass (*Zostera marina*), a type of coastal submerged aquatic vegetation, as an Ecologically Significant Species (ESS) given its contribution to and indication of coastal ecosystem health (DFO, 2009). Eelgrass was thereafter protected from any anthropogenic activity of use that would cause the harmful alteration, disruption and destruction (HADD) to eelgrass habitat (DFO, 2011a). However, recent regional eelgrass coverage declines have been observed in the sGSL (DFO, 2009; DFO, 2011a) in excess to global averages (Waycott et al., 2009), therefore suggesting that some coastal zones along the sGSL are becoming disproportionately stressed. Although five primary sources of HADD were identified (i.e., sedimentation, turbidity, nutrients, flow regime, and physical damages; DFO, 2011a), it is recognized that these stressors often occur simultaneously, and therefore the cumulative effects of multiple stressors must be considered (Bastien-Daigle et al., 2007). However, it is often difficult to disentangle cumulative effects from the effects of specific anthropogenic activities (McKindsey et al., 2006).

In regards to potential anthropogenic hazards onto eelgrass habitat, particular focus has been given to the recent and rapid expansion of the bivalve aquaculture industry throughout the sGSL and specifically the suspended bag oyster aquaculture (SBOA) practices along the eastern coast of New Brunswick (NB). Eelgrass habitat and SBOA licenses directly overlap with one another in several bays and estuaries along eastern NB (AGR, 2012; Skinner et al., 2013) with unknown SES impacts. Given that the cultured oysters are indigenous species, it is critical to consider the ecological services and functions of natural oysters in order to understand their ecological role in aquaculture (McKindsey et al., 2006). However, the interaction and relationship between oysters with the surrounding ecosystem is complex, and the net cause-effect pathways can be both positive and negative depending on the type and extent of aquacultural practices, local environmental conditions, and the spatial scale examined (Cranford et al., 2006; McKindsey et al., 2006; Bastien-Daigle et al., 2007). In the past, measures have been taken to assess the ecological impacts of SBOA on eelgrass and the greater SES along eastern NB with the purpose of sustainably developing the industry

with ICZM approaches (e.g., Transport Canada, 2007). However, DFO has recognized that the multiple overlapping human activities and resource uses (including SBOA) within the unique and complex context of the sGSL has created numerous challenges for implementing ICZM (DFO, 2005), and so the net system effects remain largely uncertain (Cranford et al., 2006).

1.3. Contextualized & Scale-Dependent Research

For the purpose of developing a holistic SES perspective and create appropriate ICZM measures, several different spatial scales were intimately examined for this research. The importance of scales and spatial heterogeneity is derived from the fact that various ecological patterns and processes operate at different scales, and that small-scale patterns cannot be simply scaled-up and extrapolated to represent large-scale areas and processes, or visa versa (Anderson et al., 2006; Schmidt et al., 2011). As such, the dominant SBOA impact onto eelgrass is entirely dependent upon the scope and spatial scale examined; a small near-field scale may magnify the impacts, while a larger estuary scale may negate other watershed-based impacts (Anderson et al., 2006). To best understand the eelgrass-SBOA relationship at multiple spatial scales, the boundaries of each scale must be explicitly defined and examined using scale-dependent methodologies since the, “question of interest must be matched to the relevant scale, and the methodologies employed must be appropriate to measurement at that scale” (Anderson et al., 2006, pg 6-7). While the distinction between different spatial scales may presently be arbitrary at best (Anderson et al., 2006), efforts have been made in this project to define the spatial scales examined in order to distinguish between the different ecological processes and patterns at the near-field, zone of influence, bay, and estuary scales. McKindsey and colleagues (2006) recommend that marine managers determine the carrying capacity limits of any given embayment in order to ensure that eelgrass health, a proxy for the functional and productive capacity of the coastal ecosystem (DFO, 2009) is maintained.

Several studies have examined the near-field effects of SBOA on eelgrass health (Skinner et al., 2013), whereas large-scale processes and far-field effects remain to be investigated (Anderson et al., 2006). It is important to note, however, that SBOA and its impacts cannot be considered completely in isolation; the multiple anthropogenic uses

and activities both within and around the estuaries cause cumulative ecosystem effects, albeit synergistic or antagonistic, with complex ecological consequences (Anderson et al., 2006). Therefore, while individual categorization and consideration has been given to each spatial scale, it is important to bear in mind that these cumulative and often cascading effects that can create long-term effects that can directly or indirectly affect the eelgrass-SBOA relationship. Hence, the literature adamantly recommends spatially explicit research in addition to a holistic ecosystem-based approach to interpreting the interactions and effects of SBOA onto eelgrass (e.g., McKindsey et al., 2006) and the greater ecosystem (e.g., Cranford et al., 2006). The holistic perspective entails incorporating the positive, neutral, and negative effects of SBOA onto eelgrass into ecosystem-based risk analyses to ensure that all environmental interactions are considered (McKindsey et al., 2006).

1.4. The Management Problem

Given that eelgrass throughout Atlantic Canada has been recognized as an ecologically significant species (ESS; DFO, 2009), the decline of meadows in the sGSL and eastern NB is both indicative as well as contributing to the degradation of ecological health the integrity of these coastal zones. However, the larger management issue is that eelgrass is intimately connected to the CRA fisheries as it provides critical nursery, refuge, and foraging habitat (DFO, 2009, 2013a), and so the ongoing decline of eelgrass is a significant risk to the CRA fisheries and can cause various direct and indirect negative socio-economic consequences (Cranford et al., 2006; McKindsey et al., 2006; Bastien-Daigle et al., 2007; DFO, 2009). Ultimately, eelgrass degradation is not only a proxy for the productivity and functionality of the ecosystem but the entire coastal SES.

Moreover, the complex multi-jurisdictional setting of the sGSL as well as existing sectoral management approaches have created an inadvertent overlap in management for coastal resource interests, uses, and activities (DFO, 2005; DFO, 2007). There is therefore an absence of definitive ecological, social, and economic objectives for the sGSL (Cormier et al., 2013). The highly complex ecosystem interactions and multiple anthropogenic pressures have made it increasingly difficult to differentiate the effects of SBOA from other coastal activities, and at which spatial scale the effect is materializing (McKindsey et al., 2006). As such, the ecological carrying capacity of

estuaries along eastern NB to accommodate the increasing amounts of SBOA activity have not been well identified or managed despite ongoing industry growth.

The management problem can be summarized as a lack of a holistic and contextualized ICZM approach for eastern NB and the sGSL, which has inevitably tolerated ongoing eelgrass degradation. Despite the various governmental documents that have mandated ICZM to be implemented in the sGSL (e.g., DFO, 2005; Transport Canada, 2007) and throughout Canada (e.g., Canada's Oceans Act, 1996; Canada's Oceans Action Plan, 2002), coastal ecosystems and resources continue to be managed with sectoral approaches. Because these legislative documents largely remain to be implemented, the prolonged sectoral approach to coastal zone management has failed to recognize the complex ecological interactions and cumulative effects of several simultaneous stressors on eelgrass habitats. As a result, SBOA management has been the product of single-species (i.e., oyster) management, and so considerations to other species (e.g. eelgrass) and the greater ecosystem have been largely excluded from management regimes.

Given that the SBOA industry and other sectors (i.e., agriculture) are recognized by the scientific community as having complex and scale-dependent effects on the coastal ecosystem (*see section 3.4.2.*), there is a significant risk for mismanagement to cause further degradation on eelgrass. Along eastern NB, the Replacement Class Screening Report (RCSR) had once offered indirect consideration to eelgrass habitat as it was considered a valued environmental and socio-economic component (VEC) (Transport Canada, 2007). However, the RCSR focused on socio-economic thresholds and failed to determine the ecological carrying capacity of SBOA in eelgrass ecosystems. Although difficult to estimate, the critical threshold at which SBOA biomass exceeds the local ECC and becomes detrimental to eelgrass productivity has not yet been quantified. As a result of ongoing sectoral management and the absence of defined ECC limits, eastern NB's eelgrass meadows will continue to degrade. The loss of eelgrass and provisioned ecological services will inevitably cause a cascade of consequences that will reduce the ecological, social, and economic benefits that are essential for human well-being (MEA, 2005; Schmidt et al., 2011). Without appropriate ICZM implementation to assess the effects of SBOA on coastal ecosystem, the health of

eelgrass habitats and the interdependent coastal SES will become increasingly vulnerable to degradation.

1.5. Research Purpose, Thesis, and Questions

The purpose of this project was to assess the effects of SBOA on eelgrass cover within the sGSL at multiple spatial scales and to conduct a holistic ecosystem-based risk management (EBRM) analysis. More specifically, the purpose of this research was to: 1) establish the scopes and effects of the eelgrass-SBOA relationship at each spatial scale; 2) assess the relationship at each of spatial scale and create a vulnerability profile by using the EBRM framework; and, 3) elaborate upon best practices and suggest ICZM recommendations that will promote the health of eelgrass and the sustainable development of SBOA along eastern NB.

It was hypothesized that each of the examined spatial scales would describe both positive and negative, direct and indirect effects of SBOA on eelgrass, and thus result in differing risk analyses outcomes across the different scales. It is further hypothesized that statistical and risk analyses would yield contextualized results specific to both the scale and embayment examined given the variety of unique hydrodynamic, environmental, and SBOA conditions within each bay and estuary. The scope and methodologies used to examine the eelgrass-SBOA relationship will therefore dictate the risk analyses' outcomes, so the ICZM recommendations will be specific to both the spatial scale and embayment examined. This research project argues that appropriate ICZM planning and implementation is necessary for DFO to achieve its mandates of sustainably developing the SBOA industry while simultaneously conserving eelgrass habitats and their ecological benefits. It is intended that the ICZM recommendations offered in this project will influence both provincial and federal legislation and policies to improve ecosystem health and support the sustainable development of the aquaculture industry in Atlantic Canada. The research questions used to help guide the project are:

- 1) Along eastern NB, how does SBOA interact with the local coastal environment, and what effects can occur on eelgrass at different spatial scales?
- 2) Which SBOA variables are most influential to eelgrass cover at the bay- and estuary-scales, and what are their critical thresholds?

- 3) What are the ecological, socio-economic, operational, and strategic consequences and overall risk likelihood at each of the spatial scales?
- 4) Which ICZM measures can be recommended in order to promote the long-term sustainable development of the SBOA industry as well as the health of coastal ecosystems along eastern NB and the sGSL?

2. Methods

2.1. Research Strategy / Methods Summary

A range of research methods was employed to address the different research questions. First, the project was limited in scope to focus on eastern NB's coastal zones with active SBOA leases (see Fig. 1 for map of bays examined). Next, two internships¹ and an extensive literature review were completed in order to collect the available information and data from a variety of sources. The review focused on the eelgrass-SBOA relationship at each of the four distinct spatial scales (see Fig 2 for example) for as many of the n=14 examined bays as possible, although occasionally lending from international examples in order to create a relative context of SBOA in NB compared to other regions.

Given that different methodologies are implicit in analyzing ecosystem relationships at different spatial scales (Anderson et al., 2006), the data were dissimilarly yet appropriately analyzed prior to the risk assessment. The near-field scale data was previously analyzed and synthesized, and did not require further analysis prior to the risk assessment. The ZOI data were analyzed using complex numerical models but required further interpretation. The far-field bay and estuary-scale data were provided raw in an Excel spreadsheet, thus requiring an array of statistical analyses and interpretation prior to their respective risk assessments.

Once the data and results were sufficiently analyzed, each of the four spatial scales was subjected to the International Council for the Exploration of the Sea (ICES) ecosystem-based risk analysis framework (Cormier et al., 2013). Using the criteria

¹ 1) Department of Fisheries and Oceans Canada (DFO) Gulf Fisheries Center in Moncton, NB, from April 29-May 17, 2013, and 2) Nova Scotia Department of Environment (NSE) Halifax, NS, from May 21-June 28, 2013.

outlined in the framework, the ecological impacts, socio-economic consequences, and operational and strategic repercussions were individually examined and ranked at each of the spatial scales. The ranking of the impacts and consequences were guided by the leading researchers and practitioners whose work directly informed this project. Then, based on the results from the ranking scores, a vulnerability profile was created for each spatial scale; the vulnerability profile in turn implied the level of tolerance for management. In cases where the results were found to not be tolerable and management actions were required, appropriate management recommendations were sought from both the literature and from the insights of experts.

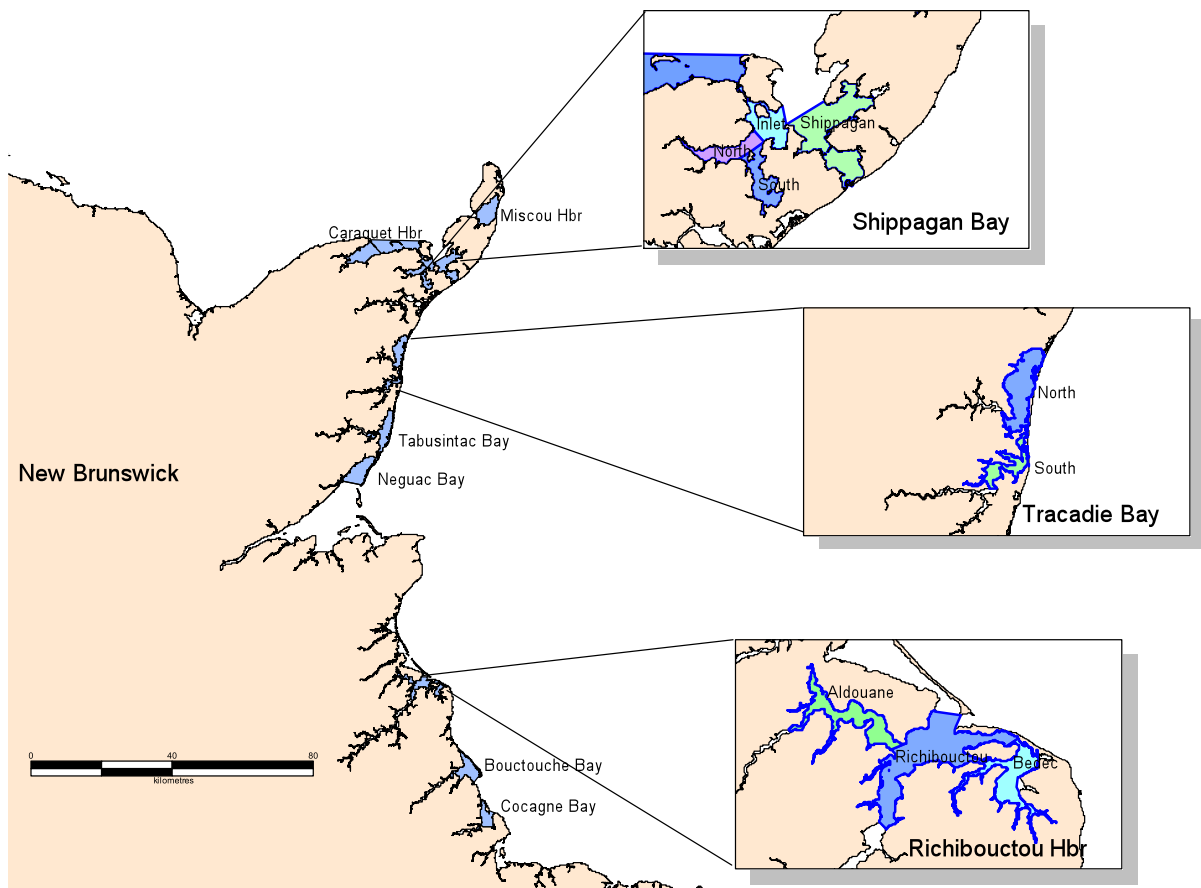


Figure 1. Map of the eastern New Brunswick coastline. The n=8 estuaries outlined in blue were examined for the purpose of this study; estuaries comprised of >1 bay were enlarged to illustrate subdivided bay boundaries. A total of n=14 bays were examined. Adapted from DFO, 2011.

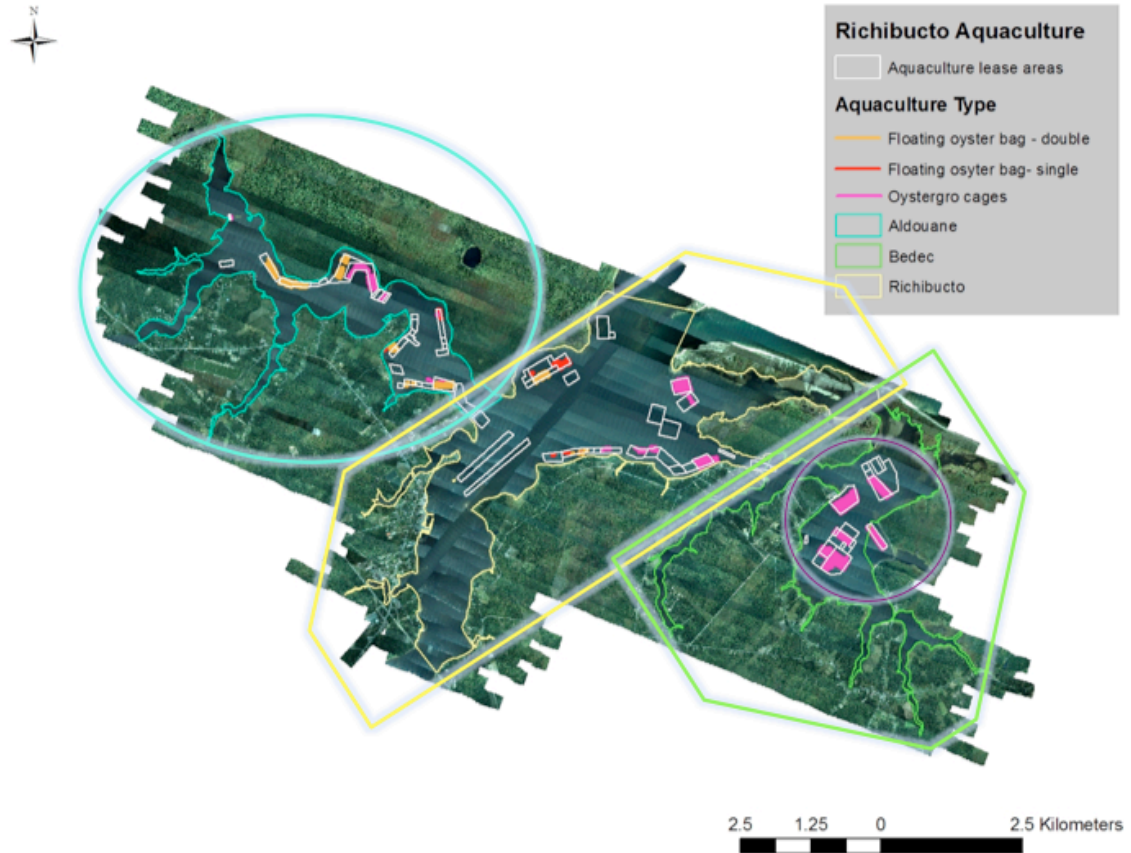


Figure 2. Map of the Richiboucto estuary illustrating the four spatial scales examined therein. The blue, yellow, and green contours describe the approximate boundaries of each of the three bays – Aldouane, Richiboucto Harbour, and Bedec, respectively – while the purple circle depicts the one of the potential zone of influence scales present within Bedec. The white outlines are the suspended bag oyster aquaculture lease areas and represent the near-field scale.

2.2. Literature review

A comprehensive literature review examined the status of scientific knowledge on eelgrass, SBOA, and their complex relationship. To the fullest extent possible, the review focused on research from eastern NB estuaries in which SBOA and eelgrass were closely examined. The review specifically examined the various biological, ecological, and socio-economic components of both eelgrass and cultured oysters, as well as their susceptibilities and pathways-of-effects. The review additionally included other local, regional, and international examples of shellfish aquaculture practices and eelgrass responses in order to contextualize the current research and results. Multiple publication sources from the Government of Canada were sourced through the DFO Waves Library webpage (www.dfo-mpo.gc.ca/waves). Academic and peer-reviewed

research was accessed via Google Scholar, Web of Science, Novanet, and Science Direct in order to efficiently peruse the available literature. Moreover, personal communication was used to access research and data that was otherwise unavailable online; contacts from DFO, Dalhousie University, Stantec, Nova Scotia Department of Environment (NSE), and others had substantiated the literature review with unpublished and in-press research.

The literature review was guided and organized according to the Marine and Coastal Ecosystem-Based Risk Management Handbook, published by the ICES Expert Group (Cormier et al., 2013). Given the time and capacity constraints associated with the current project, only the EBM context and risk identification sections of the Handbook were fulfilled and in an abbreviated format. It should be noted that applying the Handbook to multiple spatial scales is a novel application of this framework, and could serve as the foundation for future researchers hoping to further develop this technique.

2.3. Data Collection

2.3.1. Near-Field Scale Data

The near-field data used to inform the current project was collected from two primary research sources and represent the most current and explicit research available. In both cases, personal communication had led to the data dissemination. The first study was conducted by Skinner, Courtenay, and McKindsey (2013) to establish the footprint of potential near-field eelgrass effects at SBOA sites (while excluding the effects of overwintering) in five eastern NB estuaries: Bay St Simon South, Bay St Simon North, Tabusintac, Neguac, and Richiboucto estuary. The authors had statistically analyzed and synthesized their results, making it possible to directly translate the conclusions of their research into the context of this project's near-field risk assessment.

The second experiment focused almost exclusively on determining the impacts of overwintering suspended oyster bags on eelgrass beds. Conducted by Skinner, Courtenay, Boudreau, and Mallet (submitted), the experiment had tested eelgrass response to four levels of shading caused by different SBOA gear types in Bay Saint-Simon South. Although the final data collection results (May 2013) have yet to be published, the experimental design and results to date have been presented at the Ottawa

PARR meeting in April 2013 and is considered validated research (pers. comm. Monica Boudreau, May 1, 2013); the available results have contributed to the near-field risk assessment for this project.

2.3.2. Zone of Influence Scale Data

The 2013 Aquaculture Stewardship Council (ASC) Bivalve Standard inspired analyzing the effects of SBOA onto eelgrass at the zone of influence (ZOI) scale. Given the natural filtration services provisioned by bivalves, the ASC recommends measuring the total area of phytoplankton depletion, or the bivalves' ZOI, as an ecological indicator. It is suggested that the ZOI can be used as a proxy to simultaneously determine the effects of bivalve aquaculture as well as their potential to exceed the ecological carrying capacity (ECC) of their respective bay or estuary (ASC, 2013) (see Appendix 8.2.1. for descriptions of measures). The ASC (2013) suggests that the ZOI can be calculated by comparing the time taken for cultured oysters to filter and deplete the estuary of phytoplankton biomass (clearance time – CT) to the time taken for the estuary to be naturally flushed by tides and thus replenish phytoplankton biomass (replenish time – RT). The effects of SBOA at the ZOI scale were quantified in order to determine whether these aquaculture sites are sustainable and within the natural carrying capacity limits of the embayment.

Most recently, Guyondet, Sonier, and Comeau (2013) have developed a numerical model to represent the spatial and temporal dynamics of phytoplankton availability (using CT, RT, and PPT) throughout the Richiboucto estuary and its three subsidiary bays (Aldouane, Richiboucto Harbour, and Bedec) in order to determine the seston depletion caused by SBOA. During an internship at DFO Gulf Fisheries Center (Moncton, NB), the models were manipulated and applied to test different theories of seston depletion caused by SBOA; these results are specific to the current project and are not included in the Guyondet and colleagues (2013) publication. First, because seston depletion was not monitored prior to the installment of SBOA, a phytoplankton biomass baseline was modeled by removing the influence of CT throughout the entire estuary (i.e., total absence of SBOA in Richiboucto). Next, the model had simulated the degree of seston depletion for the entire estuary when CT was restored to only one bay at a time. Aldouane and Bedec were individually modeled to have the only CT present

in the Richiboucto estuary by assuming the absence of SBOA sites elsewhere. Lastly, because Bedec is a relatively isolated and enclosed bay, the models were able to calculate the total area and proportion of the bay that corresponded to an SDI score <0 (i.e., phytoplankton depletion relative to sGSL concentrations). The ZOI model outputs were examined in the context of the available literature to infer the effects and risks of SBOA on eelgrass at the ZOI scale.

2.3.3. Bay, Estuary, and Watershed Scale Data

For the purpose of the current research project, the bay- and estuary-scales are defined by the physical boundaries of each bay and estuary, respectively, with watershed land-uses influencing the respective estuary and therefore considered a part of the estuary-scale (see Appendix 8.2.2 for figures). The datasets used to represent the eight estuaries and 14 bays along eastern NB were sourced from a variety of primary governmental research projects and represent the most accurate ground-truth data available. Although much of the research remains to be published, the data was provided by Environment Canada (EC), Fisheries and Oceans Canada (DFO), NB Department of Agriculture, Aquaculture, and Fisheries NB (NB AAF), and Natural Resources Canada (NRC). The data had been communicated to and amalgamated by the DFO Gulf Fisheries Center (Moncton, NB) Science Branch prior to having been provided in a Microsoft Excel spreadsheet. Given that the data had been sourced from a number of governmental agencies, it had first been communicated to M. Niles (DFO, Gulf Fisheries Centre) prior to being amalgamated and presented in a Microsoft Excel spreadsheet for analyses.

Environment Canada had most recently calculated the total area of each of the eight estuaries and 14 bays from 2007-2009 using a variety of methods, including aerial photography and satellite imagery (pers. comm. Monique Niles, April 29, 2013). Using aerial photography, Quickbird satellite imagery, side-scan sonar, video camera transects and/or drop camera methods, EC had used computer processing to detect eelgrass presence in the bays and estuaries from 2007-2009. The EC results were compared to the ground-truth data compiled by DFO's field surveys for the same years, and together the EC and DFO results created eelgrass cover estimates for each of the eight estuaries

and 14 bays (pers. comm. Monique Niles, October 7, 2013) (see Appendix 10.2.1. for summary table of EC/DFO data collection methods per estuary).

Aquaculture data is annually collected in accordance with both provincial and federal legislation. NB AAF biologists had conducted field surveys to record active and total SBOA lease area (in hectares), total aquaculture lease area (sum of total suspended and bottom lease area), gear types, and SBOA bag counts for all registered SBOA sites. SBOA bag count data were collected for the same years that the estuaries were sampled for eelgrass (e.g., 2007 SBOA bag counts were amalgamated for Richiboucto; 2009 bag counts for Neguac). Active SBOA, total SBOA, total aquaculture lease, and gear area data were unavailable for the same time periods of eelgrass data collection; as a result, 2011 field survey data were used to inform the datasets² with high DFO confidence that lease areas had not significantly changes from 2007-2009 to 2011 (pers. comm. Monique Niles, October 9, 2013). Using the NB AAF field survey results, DFO had estimated total aquaculture biomass per bay and estuary (in tonnes) by reasoning that each gear type had a standard line length and that all lines have a standard biomass of 6.04 kg/bag (Comeau, 2006). Moreover, A. Locke (DFO, unpublished data) had converted the total aquaculture biomass per bay area into dry tissue weight (g per m²) in order to better compare results to those of Newell and Koch (2004) for an international context. Gear area was estimated as the product of the known dimensions of each gear type and number of gear units (pers. comm. Monique Niles, April 29, 2013). Again, all aquaculture data was first communicated to M. Niles (DFO, Gulf Fisheries Centre) prior to being amalgamated and presented in a Microsoft Excel spreadsheet for analyses.

Watershed area³ for each estuary was acquired from NRC as well as the area of forested and non-forested land-use per watershed. Non-forested area was further described according to seven primary land-uses; agriculture (AGR), Department of National Defense (DND), industrial (IND), infrastructure (INF), recreational (REC),

² With the exception of Miscou, for which 2009 active SBOA lease area was provided given that there was no recorded aquaculture in 2011; however, gear area (2011 data) remains to report 0 ha due to this discrepancy.

³ The various land uses surrounding the estuaries within each respective watershed have the propensity to influence the eelgrass-SBOA relationship via the addition of land-based nutrients from run-off that could promote phytoplankton production and in turn support oyster populations above the natural ECC.

settlement (SET), and wilderness (WIL). The NRC data was provided in ArcGIS shapefiles that were interpreted by M. Niles by having measured the area of each land-use and presenting the results in a Microsoft Excel spreadsheet (see Appendix 8.2.2. for a map of NB watersheds and definitions of each primary land-use).

2.4. Data Analyses

2.4.1. Datasets

Two datasets were created to distinguish between individual bays and entire estuaries in order to eliminate the high correlations amongst estuaries and their contributory bays, which would have otherwise violated regression assumptions⁴. Where all of the bays within a respective estuary were surveyed, the variables for each bay were added to create estuary-scale data (e.g., Aldouane + Richiboucto Harbour + Bedec = Richiboucto estuary). For the embayments that were measured as a single unit and could not be differentiated as either a bay or estuary (e.g., Bouctouche), the basin was included in both bay- and estuary-scale analyses. In the case of Shippagan estuary in which only one of its three contributory bays was completely surveyed (i.e., Shippagan South) only the one bay was kept in the dataset and the estuary was omitted. Similarly, only the northern half of Tracadie estuary (i.e., Tracadie North) was completely surveyed while the southern bay had missing eelgrass cover data; eelgrass cover was later estimated, allowing for both bays to be included in the bay-scale dataset and the estuary in the estuary-scale dataset.

Watershed variables were included in the estuary dataset given that there is incomplete information as to the definite end-point of land-based run-off within the estuary; that is, whether there is one bay that receives the majority of the run-off in comparison to its adjacent bay. As a result of the inability to differentiate the watershed effects at the bay scale, it is assumed that a watershed's land-uses affects its respective estuary ubiquitously. The final datasets were comprised of n=14 bays and n=8 estuaries and are summarized below.

⁴ By distinguishing bays from estuaries, the regression assumptions of no correlation amongst independent variables were upheld and prevented skewed results (pers. comm. Stu Carson, June 17, 2013).

Table 1. List of the eastern New Brunswick estuaries (n=8) and bays (n=14) used for statistical analyses.

Estuary	Bay
1. Miscou	1. Miscou
2. St-Simon	2. St-Simon Inlet
	3. St-Simon North
	4. St-Simon South
Shippagan*	5. Shippagan South
3. Tracadie	6. Tracadie North
	7. Tracadie South
4. Tabusintac	8. Tabusintac
5. Neguac	9. Neguac
6. Richiboucto	10. Aldouane
	11. Richiboucto Harbour
	12. Bedec
7. Bouctouche	13. Bouctouche
8. Cocagne	14. Cocagne

*Note: Shippagan was omitted from analyses as an estuary but one of its constituent bays, Shippagan South, was included in bay-scale analyses.

Eelgrass cover (ha) per bay and per estuary was the dependent variable for all appropriate subsequent analyses. The data were cleaned by having removed significant correlations (>0.60) and the remaining data was used as the raw independent co-variables (see Appendix 10.2.3. for greater description and list of data). Bay-scale independent were multiplied by bay area to create relative interaction terms⁵ at the bay-scale; there were not sufficient degrees of freedom to allow interaction terms at the estuary-scale (see Table 2 below for list). In order to generate spatially-realistic results, the dependent and independent variables were appropriately matched according to spatial scales (e.g., eelgrass cover per bay was tested against bay-scale co-variables, eelgrass cover at the estuary scale was tested against estuary independent variables). Restricting the statistical analyses to appropriate scales allowed for spatially-unique processes to influence the results. At the bay scale, small differences amongst hydrodynamic and oceanographic processes are likely to be more influential on the

⁵ Interaction terms account for independent variables that do interact such that effect on the dependent variable (i.e., eelgrass cover) of one independent variable depends on the value of the other (e.g., active SBOA lease area is a function of the bay area). The resulting effects are not additive, and are therefore interactive (pers. comm. Joey Hartling, September 20, 2013).

results whereas these processes may have become counterbalanced at the estuary scale⁶. Watershed variables were analyzed in the absence of a dependent variable given that the correlations were too high for regression analyses, and were consequently limited to principle components analyses. A summary of the bay, estuary, and watershed-scale variables, including both raw and interaction terms, were summarized in Appendix 8.2.3.

2.4.2. Statistical Tests

All multivariate comparisons of eelgrass cover and SBOA variables were tested using multiple step-wise regressions, principle component analyses, and sensitivity analyses. All statistical analyses were performed using RStudio (RStudio Integrated Development Environment, Massachusetts, USA). It was assumed that the remaining n=11 bay and n=12 estuary-scale (n=6 estuary and n=6 watershed) co-variables (see Table 2 below for list) were independent and assumptions of normality were achieved upon removing high correlations (>0.60) and insufficient datasets (see Appendix 8.2.3. for descriptions).

Table 2. List of dependent and independent variables used for statistical analyses at the bay (n=11; n=6 raw data and n=5 interaction terms), estuary (n=6), and watershed (n=6) scales. Abbreviations and expressed units for the variables are in parentheses.

	Bay-Scale Variables	Estuary-Scale Variables	Watershed-Scale Variables
Dependent Variable	Eelgrass cover (ha)	Eelgrass cover (ha)	/
Independent Raw Co-Variables	Active SBOA lease (ASL) area (ha)	Active SBOA lease (ASL) area (ha)	Agriculture area (AGR) (ha)
	Total SBOA lease (TSL) area (ha)	Total SBOA lease (TSL) area (ha)	Infrastructure area (INF) (ha)
	Total aquaculture lease (TAL) area (ha)	Total aquaculture lease (TAL) area (ha)	Industrial area (IND) (ha)
	Bag counts	Bag counts	Settlement area (SET) (ha)
	Gear area (ha)	Gear area (ha)	Forest area (ha)
	Bay area (ha)	Estuary area (ha)	Watershed area (ha)
Independent Interaction Terms	ASL * Bay area		
	TSL * Bay area		
	TAL * Bay area		
	Bag count * Bay area		
	Gear area * Bay area		

⁶ For example, tidal residency time in Bedec is significantly longer than that of Aldouane or Richiboucto Harbour, causing significant differences in seston depletion rates that in turn would effect the ECC of oyster biomass; however, the tidal residency time and its effects are averaged for the three bays at the estuary scale.

The statistical analyses were performed on the raw bay- and estuary-scale data. Upon removing the SBOA co-variables with high correlations, multiple step-wise regression analyses (MRAs) were conducted to generate non-linear models in order to determine the most significant co-variables. Then, the raw data was transformed into z-scores prior to performing Principal Component analyses (PCAs) to determine which co-variables most accounted for the variance explained. Sensitivity analyses (SAs) were developed specifically for the current research project to predict eelgrass response to different SBOA conditions (see Appendix 8.2.3.1-8.2.3.3 for greater descriptions of each test). The MRAs, PCAs, SAs were each conducted using the bay and estuary data while watershed-scale co-variables were only analyzed using the PCA. The results were interpreted to help explain the eelgrass-SBOA relationship at the bay and estuary scales.

2.4.2.1. Eelgrass Cover Prediction Analysis

Prior to performing statistical tests, it was necessary to predict the eelgrass cover for Tracadie South in order to increase the statistical power and degrees of freedom of the subsequent analyses. A prediction function was employed using a best-fit model onto the entire bay-scale ground-truth dataset in order to predict for eelgrass (see Appendix 8.2.3.1. for description of model; see Table 3 below for results). The model was found to be highly predictive of eelgrass cover with a mean difference of 4.44% between the ground-truth data and the predicted results; however, the model had underperformed for two bays (>16% variance), indicating that the predictions remain to be imperfect despite being the best fitted model possible.

Although the predicted eelgrass cover for Tracadie South (1944 ha) was found to be greater than its bay area (971 ha), this spatial misrepresentation should be considered as a valid contribution to the model given the model's overall validity despite this significant outlier. The predicted cover was then added to the EC/DFO eelgrass cover for Tracadie North (1087 ha) to create an estuary-scale eelgrass cover estimate for Tracadie estuary (3031 ha). With the exception of the two predicted values for Tracadie South bay and Tracadie estuary, the ground-truth data was kept as the response variable in all subsequent analyses. It should be noted, however, that the Tracadie South eelgrass predictions (see section 2.4.2.5.) were discarded from bays-scale synthesis due to the consistent over-estimates of eelgrass per bay area (e.g., approximately 200%),

suggesting that while the model had well predicted the eelgrass cover for bays with ground-truth data, it had failed to accurately predict eelgrass cover Tracadie South. Interestingly, the Tracadie estuary predicted eelgrass estimates were found within a reasonable range of expected results (e.g., <100% eelgrass cover per bay area). Ultimately, these predicted values added a degree of freedom to each the bay and estuary-scale datasets, therefore improving the statistical power of all subsequent analyses.

Table 3. Area of eelgrass cover (ha) per bay (n=14) from EC/DFO data and the GLM model predictions. The % difference shows the difference of the eelgrass cover predictions relative to the EC/DFO estimates.

Bay	EC/DFO Eelgrass Cover (ha)	Predicted Eelgrass Cover (ha)	% Difference
Miscou	2234	2226	0.36
St-Simon Inlet	648	756	16.67
St-Simon North	613	515	15.99
St-Simon South	668	660	1.20
Shippagan South	761	770	1.18
Tracadie North	1087	1076	1.01
Tracadie South	/	1944	/
Tabusintac	1326	1299	2.04
Neguac	1875	1882	0.37
Aldouane	440	436	0.91
Richiboucto Hbr.	1030	1081	4.95
Bedec	467	472	1.07
Bouctouche	1719	1705	0.81
Cocagne	935	956	2.25

2.4.2.2. Linear & Multiple Step-Wise Regressions

Linear regressions were performed to examine the direct relationships between eelgrass cover and each of the n=11 bay-scale and n=6 estuary-scale SBOA co-variables. The lack of significance for the majority of co-variables as well as the non-linear distribution of the data had been the basis for assuming a non-linear relationship between eelgrass cover and the SBOA co-variables. As a result, the multiple regression analyses (MRA) assumed non-linear data distributions in order to establish the best-fit model.

A backwards MRA was performed on the bay and estuary-scale datasets. The MRA focuses on the relationship between the dependent variable (i.e., eelgrass cover) and all n=11 bays-scale and n=6 estuary-scale independent variables as they interact together in the model. The MRA begins with the full dataset as the null model and

sequentially removes the least significant independent variables until the quality of the remaining statistical model cannot be improved (i.e., a lower AIC score and higher R^2 value cannot be achieved). Additionally, the MRA can reveal which independent variables contribute positively or negatively to the model, and in turn the response of eelgrass cover at the bay or estuary scale. The bay and estuary-scale datasets were each tested with the MRA in order to determine 1) which SBOA variables most influence eelgrass cover at each spatial scale, and 2) how do these SBOA variables differ and compare across spatial scales. Both the bay and estuary-scale MRA models yielded residuals against fitted values (RAFV) plot and a normal Q-Q plot. The MRA results were used to inform the risk analysis.

2.4.2.3. Principal Components Analysis

A PCA was performed in order to transform the dataset into a more manageable representation by rotating the data into a new configuration for simpler interpretation. The newly configured data is best explained by new axes, or dimensions, from which relationships of the co-variables and patterns in the units become illustrated. The $n=6$ bay, estuary, and watershed raw independent variables⁷ were first transformed into z-scores in order to standardize the data prior to each PCA given that PCA is sensitive to the relative scaling of the data (pers. comm. Joey Hartling October 9, 2013). The $n=6$ z-scores were then compiled into a data frame that was then fed into a PCA model. Because the data was standardized (i.e., mean = 0) yet the variables are in different units, the analyses were based on the data's correlations matrix. The bay, estuary, and watershed-scale datasets were each tested with PCAs in order to determine 1) which co-variables are most similarly associated, and 2) how do these associations compare across the spatial scales. These results were used to inform the risk analyses.

2.4.2.5. Sensitivity Analysis

A sensitivity analysis (SA) was designed specifically for the purpose of the current project in order to determine eelgrass cover trends for each bay and estuary. The SA was designed to determine the effects on eelgrass cover in response to the theoretical

⁷ Despite gear having been dropped in the MRA, the model had incorporated gear data in the estuary-scale PCA given that the model is robust to high correlations.

increase or decrease of a single SBOA variable by keeping all other co-variables constant (see Appendix 8.2.3.3. for description of the model). Results were normalized by having divided the predicted eelgrass cover by its respective basin area, then compared to and plotted against percent EC/DFO eelgrass cover per basin area to create the mean percent difference in eelgrass cover to provide information as to the mean growth and/or decline in predicted eelgrass cover in response to an increase or decrease in the SBOA values. Once plotted, lines of moving averages (LMAs) were used to help simplify the visual interpretation of the results in order to determine the presence of relationships between eelgrass and the manipulated co-variables, as well as any bay- or estuary-scale patterns⁸.

It was hypothesized that relative to the MRA predictions, a decline in eelgrass cover in response to a maximum variable (or the increase in predicted eelgrass in response to a minimum variable) would be indicative of the basin being at or beyond the local natural ECC limits and would require immediate SBOA management treatment. Inversely, it was hypothesized that a relative increase in eelgrass cover in response to a maximum variable (or the decline in predicted eelgrass cover in response to a minimum variable) would be indicative of the basin being within natural ECC limits and could potentially be further optimized with management (see Appendix 10.2.3.3. for summary table). The resulting bay- and estuary-scale trends informed the risk analysis as well as the SBOA management recommendations.

2.5. Risk Analyses

The Risk Management Tools framework (IOM Practitioners Workshop, 2013) was used as the central framework to structure the risk vulnerability analyses of the eelgrass-SBOA relationship at multiple spatial scales. This framework represents a generic yet structured method to apply an EBRM approach specifically for marine and coastal ecosystems by providing standardized criteria for risk management impacts and likelihood assessments. The risk management criteria used to assess the degree of impacts and consequences of the risk is scaled 1-5 such that: 1) negligible; 2) low; 3)

⁸ Plots per bay and per estuary depicting the response to each of the manipulated co-variables were considered but not included given the known imperfection of the model, to which individual bay inferences may be beyond the power of the statistics used and could misrepresent the results. Instead, plots depicting the bay-scale trends in response to manipulated co-variables were presented.

medium; 4) very high; and, 5) extreme. The risk management likelihood criteria used to assess the likelihood of a risk occurring is scaled from 1-5 such that: 1) rare (<5%); 2) unlikely (5-24%); 3) moderate (25-75); 4) likely (76-95%); and, 5) almost certain (>95%). The risk management impact criteria were used to assess the ecological impacts, socio-economic consequences, and operational and strategic repercussions (see Appendix 8.2.4. for criteria and likelihood descriptors).

The four spatial scales of the eelgrass-SBOA relationship were individually analyzed using the aforementioned framework. Upon determining the level of impact and likelihood associated with each spatial scale, a risk analysis “heat map” was generated according to the IOM framework to summarize the overall degree of risk to management; the heat map in turn can be used to infer the level of tolerability that risk presents to managers and decision-makers (see Appendix 8.2.4. for examples). For risks that are not tolerable, an ICZM recommendation was provided. It should be noted that the risk analyses are contextualized to both the scale and sample sites considered, and the results should not be directly extrapolated onto other scales and sites.

2.6. Management Options and Recommendations

Based upon the available research as well as the novel results presented in the current project, a series of EBRM recommendations were developed to support the longevity of estuarine health and integrity as well as the sustainable development of the SBOA industry in eastern NB. These recommendations were supplemented with professional insights from experienced researchers.

3. Literature Review

3.1. The southern Gulf of St Lawrence and eastern NB: The Social-Ecological Unit

Atlantic Canadian coastal waters encompass a variety of coastal and marine ecosystems; in particular, the Gulf of St Lawrence has been federally recognized as having a unique and complex SES (DFO, 2005). Moreover, the southern Gulf of St Lawrence (sGSL) is considered a distinct SES within the Gulf given its enhanced complexity and productivity given that it provides important spawning, nursery, and adult feeding habitats for several fish stocks (DFO, 2005) including commercial, recreational, and aboriginal (CRA) valuable fisheries (Kenchington et al., 2012). As

such, the sGSL is regarded as having distinct food web and maintains a high degree of marine biodiversity given the large increase in seasonal plankton biomass, the highest in Atlantic Canada (Bastien-Daigle et al., 2007; DFO, 2005). Among the five provinces bordering the sGSL⁹, the eastern NB coastline (see Figure 3 below for map) and its estuaries significantly contribute to the overall biodiversity and productivity in the sGSL (Bastien-Daigle et al., 2007; Turcotte –Lanteign and Ferguson, 2008a).

The relatively high ecological productivity and coastal resources provisioned by the sGSL and eastern NB's unique ecosystems have supported and shaped a variety of human activities, industries, and livelihoods throughout Atlantic Canada (DFO, 2005) (e.g., CRA fisheries, aquaculture). The sGSL is considered a culturally and socially distinct and complex system given that each of the three surrounding provinces¹⁰ possess their own variety of heritages and cultures (DFO, 2005) that inevitably influence the local and regional SES. However, these activities can exacerbate the vulnerability of these coastal ecosystems and threaten the same resources that the social and economic systems are dependent upon (Bastien-Daigle et al., 2007; Turcotte-Lanteign and Ferguson, 2008a). In the case of eastern NB, both eelgrass and oyster aquaculture are prominent features that often spatially overlap (AGRG, 2012; Skinner et al., 2013), resulting in ecological and socio-economic benefits as well as consequences.

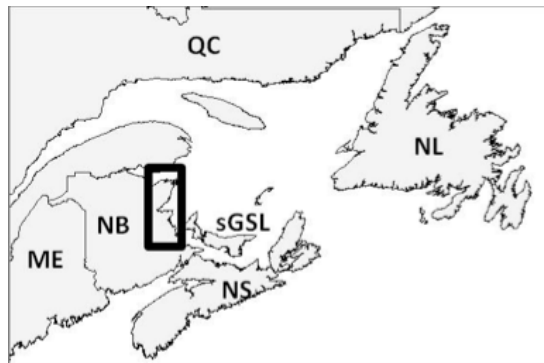


Figure 3. Map of Atlantic Canada with focus on the eastern New Brunswick (NB) coastline along the southern Gulf of St Lawrence (sGSL). Adapted from Skinner et al., 2013.

⁹ The Gulf of St Lawrence is surrounded by Quebec (QC), Newfoundland and Labrador (NFL), Prince Edward Island (PEI), Nova Scotia (NS), and New Brunswick (NB).

¹⁰ The sGSL is comprised of the bays, inlets and estuaries of PEI, the eastern coast of New Brunswick, and the northern shore of Nova Scotia.

3.2. Valuable Ecosystem Components and Services

3.2.1. Eelgrass

It has been formally recognized that there is a link between coastal ecosystem health and eelgrass abundance; as a result, eelgrass in Atlantic Canada has been considered an Ecologically Significant Species (ESS) given that it is a critical indicator and contributor to coastal ecosystem health and thus the regional SES (DFO, 2009). Essentially, an abundance of eelgrass is indicative of a healthy coastal ecosystem whereas a stressed ecosystem will have declining or extirpated eelgrass cover (AGRGR, 2012). Despite the small surface area and biomass that eelgrass and other seagrasses represent on the global scale (coverage is <0.2% of the global oceans), the relatively high value of their ecosystem services and functions make eelgrass disproportionately valuable habitats in comparison to other ecosystems (Costanza et al., 1997; Schmidt et al., 2011; Fourqurean et al., 2012). Eelgrass provides structural complexity to the benthic zone, enhances food resources and habitat surface area, and therefore plays an important role as a near-shore spawning, nursery, refuge, and adult feeding habitat for a variety of fish and invertebrate species (Locke and Hanson, 2004; Vandermeulen, 2005; DFO, 2009). Eelgrass and other seagrass meadows are additionally renowned for their role in climate regulation, nutrient cycling and other services (i.e., absorbing nutrients, storing organic matter, sequestering carbon, stabilizing suspended sediments, buffering shorelines from erosion, increasing light attenuation, filtering contaminants, and producing dissolved oxygen) (DFO, 2009; Vandermeulen, 2009; Fourqurean et al., 2012) that can directly benefit socio-economic systems. It is widely recognized that there is no functional substitute for eelgrass in the sGSL given that no other structural habitats or species (e.g., oyster beds, macroalgae) can provide these ecological services and functions to the extent that eelgrass does; in the absence of eelgrass, the benthic zone would consist largely of sand and/or mud flats (DFO, 2009; DFO, 2011).

3.2.2. American Oysters

The primary bivalve species cultured along eastern NB is the American oyster (*Crassostrea virginica*) (McKindsey et al., 2006). Oyster farming began in NB in 1865 with collected natural seed transplanted onto bottom lease areas (Comeau, 2013); however, this traditional approach has progressively been replaced with suspended bag

oyster aquaculture (SBOA) practices and is now the dominant aquaculture method in NB (Comeau, 2013; Skinner et al., 2013). In 2012, NB produced 1,118 tonnes (t) of cultured oysters alone, worth approximately \$5.2 million; this estimate is almost double what DFO reported the year before, with only 609 t of oysters harvested in 2011 worth \$2.65 million (DFO, 2013b)¹¹ (see Fig 4 for 1996-2009 estimated bag equivalents for eastern NB). Indeed, the industry has grown considerably in the past 25 years (i.e., approximate 7.5-fold increase in harvest and value since 1986) and further expansion is expected in the future (Mallet et al., 2006; Bastien-Daigle et al., 2007; Comeau, 2013; Skinner et al., 2013).

It is important to make the distinction that oyster aquaculture in NB and throughout Atlantic Canada is considerably less intense than in other areas throughout the world¹² and therefore constitutes a relatively low-intensity production by comparison (Comeau et al., 2006; Bastien-Daigle et al., 2007; Skinner et al., 2013). Moreover, although there is a historically and naturally high carrying capacity and dominance of oysters in regional coastal ecosystems (Bastien-Daigle et al 2007), natural oyster stocks (i.e., non-cultured) have strongly declined and are nearly negligible throughout the sGSL (Comeau, 2006) and along the eastern North American coastline (Newell and Koch, 2004). Therefore, cultured oysters represent the majority of remaining oyster populations along eastern NB and the sGSL (Bastien-Daigle et al., 2007).

¹¹ It should be noted that there is a significant difference between the annual amount of cultured oyster bags and biomass reported within an embayment and the amount harvested. Oysters take years to grow to market size (3-5 years depending on gear type) and although juvenile oysters are not economically valuable, they are ecologically significant.

¹² Estimated 2-4 t ha⁻¹ yr⁻¹ in eastern NB (Bastien-Daigle et al., 2007) is considerably less than production sites in France (8.00 t ha⁻¹ yr⁻¹ in Thau Lagoon; De Casabianca et al. 1997) or USA (13.36 t ha⁻¹ yr⁻¹ in Toten Inlet, WA; Brooks 2000). Comeau and colleagues (2006) make the similar comparison of 0.23 kg m⁻² of leased area in NB compared to 10 – 85 kg m⁻² in other areas in the world.

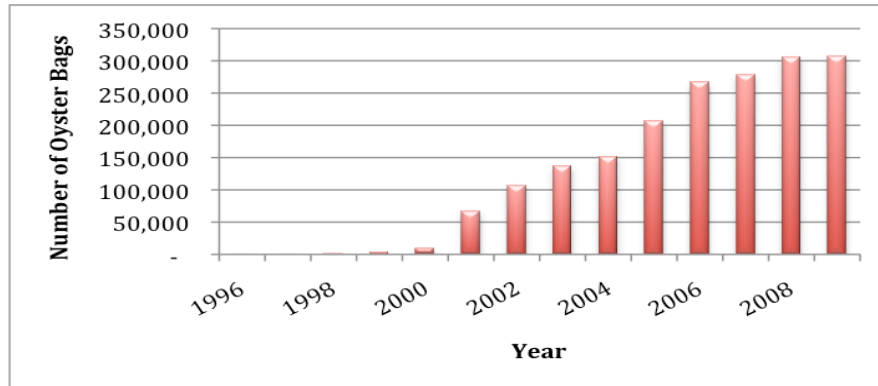


Figure 4. Number of annual oyster aquaculture bag equivalents reported from 1996-2009 along NB's eastern shoreline. Adapted from Comeau, 2013.

Regardless of population size, cultured and natural oysters fulfill many of the same ecological roles, services, and functions, although with differing habitat effects (McKindsey et al., 2006; Bastien-Daigle et al., 2007). Throughout the literature, it is agreed that SBOA has the propensity to significantly affect their surrounding environmental conditions¹³, resource availability, and species interactions, either directly or indirectly, positively or negatively (McKindsey et al., 2006; Bastien-Daigle et al., 2007). It has become increasingly recognized that bivalves are an essential coastal ecosystem component given the production of necessary ecological services and functions that promote the sustainability of the ecosystem (Prins et al., 1997), provide socio-economic benefits (Bastien-Daigle et al., 2007), and increase the resilience of the coastal SES (Bastien-Daigle et al., 2007).

Through filter feeding, oysters are known to affect system productivity, water clarity, nutrient cycling and dynamics, and coastal food webs. Oysters affects the coastal food web by being able to influence biomass and species composition of plankton communities (mitigate eutrophication), remove inorganic matter (reduce turbidity, improve water clarity, improve light attenuation), and biodeposition, which can have cascading effects on the entire SES¹⁴. Phytoplankton regulation is among the oysters' most valued provisioned services given that their physiological plasticity allows them to increase filtration rates in response to increased phytoplankton abundances (Newell

¹³ SBOA gear can act as artificial breakwaters that can reduce wave energy, sediment re-suspension, and shoreline erosion (McCormick-Ray 1998, as cited by Newell and Koch, 2004).

¹⁴ See McKindsey et al., 2006 and Champlain et al., 2006 for a review on the ecological functions of wild and cultured bivalves.

2004, as cited by McKindsey et al., 2006). Oysters can optimize primary production through selective grazing (Newell and Koch, 2004) and grazing of older plankton cells, potentially allowing for a shift to faster-growing phytoplankton species (Prins et al. 1995). As a result, oysters can mitigate the effects of eutrophication (e.g., Anderson et al., 2006; Dame, 1996; Landry, 2002; Newell and Koch, 2004), which in turn can enhance production of eelgrass (Newell and Koch, 2004) and thus have positive SES effects. Because of their pronounced influence on the coastal systems, oysters have been termed as keystone meta-populations, foundation species, ecosystem engineers, and biogenic habitats, among other terms (reviewed by McKindsey et al., 2006 and Basiten-Daigle et al., 2007).

3.3. Legislation, Policies, Management Outcomes, and Governance Structures

Due to the complexity of the GSL and its unique SES, the area has been established as a Large Oceans Management Areas (LOMAs) and is formally subjected to “integrated oceans management for sustainable development” (DFO, 2005) under Canada’s Oceans Action Plan (2002) and overarching Oceans Act (1996). Furthermore, the sGSL was distinctively recognized as one of the Ecologically and Biologically Significant Areas (EBSA) since 2007 for its rare biodiversity and productivity (DFO, 2007). Upon further investigation of the coastal ecosystems, which were excluded for review in the designation of the EBSA (DFO, 2007), it was found that eelgrass met the criteria to be designated as an Ecologically Significant Species (ESS) given the disproportionate value of its ecosystem services and functions that ultimately benefit the greater SES (DFO, 2009). Under the Canadian Environmental Assessment Act (CEAA) (1992) and old Fisheries Act (1986), eelgrass meadows were considered as critical fish habitat and were indirectly protected from HADD unless authorized under Section 35 of the Fisheries Act (DFO, 2009; DFO, 2011). Upon having assessed all potential sources of HADD onto Atlantic Canadian eelgrass (DFO, 2011a), Transport Canada and other governmental agencies (2007) developed a Replacement Class Screening Report (RCSR) for water column oyster aquaculture in NB to prevent further harmful effects onto the services and functions provisioned by eelgrass as fish habitat. The RCSR had catalyzed a series of regional workshops to discuss SBOA planning requirements and

found that the socio-economic threshold of risk tolerability would not allow >10% of the surface area of a bay to be reserved for SBOA (DFO, 2011b).

However, recent changes to the Fisheries Act (2013) have dramatically changed the terminology such that only, “serious harm to fish that are part of a [CRA] fishery, or to fish that support such a fishery” are prohibited. Although eelgrass and other plant-based biogenic habitats have demonstrated links to the productivity of CRA fishery species (Kenchington et al., 2013), they do not meet the new legal definition of “fish” and are therefore not currently mandate for protection (DFO, 2013a). However, recent reviews by DFO have recognized submerged aquatic vegetation (e.g., eelgrass) as critical fish habitat given its supporting structures and functions that directly and indirectly link to the CRA fisheries (Kenchington et al., 2013), and therefore merits consideration under the Fisheries Act (Koops et al., 2013).

Moreover, the recently reduced capacity of the government to conduct Environmental Impact Assessments (EIAs) has resulted in the termination of the RCSR projects and has allowed for less stringent environmental monitoring and assessment of effects from oyster aquaculture in NB. Analyses of the SBOA lease area per bay area found that four of the n=14 bays examined along eastern NB (i.e., St Simon North and South, Aldouane, and Bedec) have SBOA lease area >10% of bay area and are therefore beyond the agreed socio-economic threshold (DFO, 2011b). Lastly, Canada has not described any ecosystem management outcomes¹⁵ expected from the current or future implementation of the RCSR, Fisheries Act, or any other policies (Cormier et al., 2013). Without clear management objectives, Canada and its provinces will likely be unable to achieve the sustainable development of the estuarine, coastal, and marine ecosystems and its industries.

There are multiple complex property and jurisdictional overlaps and issues with respect to using Atlantic Canadian waters for aquaculture (Fig. 5). Despite eelgrass and oyster aquaculture co-existing in the area and influence one another’s services and functions, they are subjected to different regulatory agencies and thus jurisdictional boundaries (i.e., DFO for eelgrass, NBAAF for NB oyster aquaculture). Although the

¹⁵ Ecosystem management outcomes describe the expected results of existing or future implemented management strategies (Cormier et al., 2013).

federal government is the primary stakeholder, manager, and decision-maker in the integrated management of Canadian coastal zones (given that they are responsible for developing and implementing the regulations and acts used to manage these zones), much of the management is being done at the community level where people bear the brunt of poor management outcomes (Turcotte-Lanteign and Ferguson, 2008b). Many community groups in eastern NB are confused by the various roles and responsibilities of different government levels and agencies, and their respective jurisdictional boundaries (Turcotte-Lanteign and Ferguson, 2008b). Lastly, First Nation people enact their own marine governance regimes; those who live on a reserve are exempt from provincial regulations and laws, but are only exempted from federal laws if it is included in an ancestral treaty (Turcotte-Lanteign and Ferguson, 2008b).

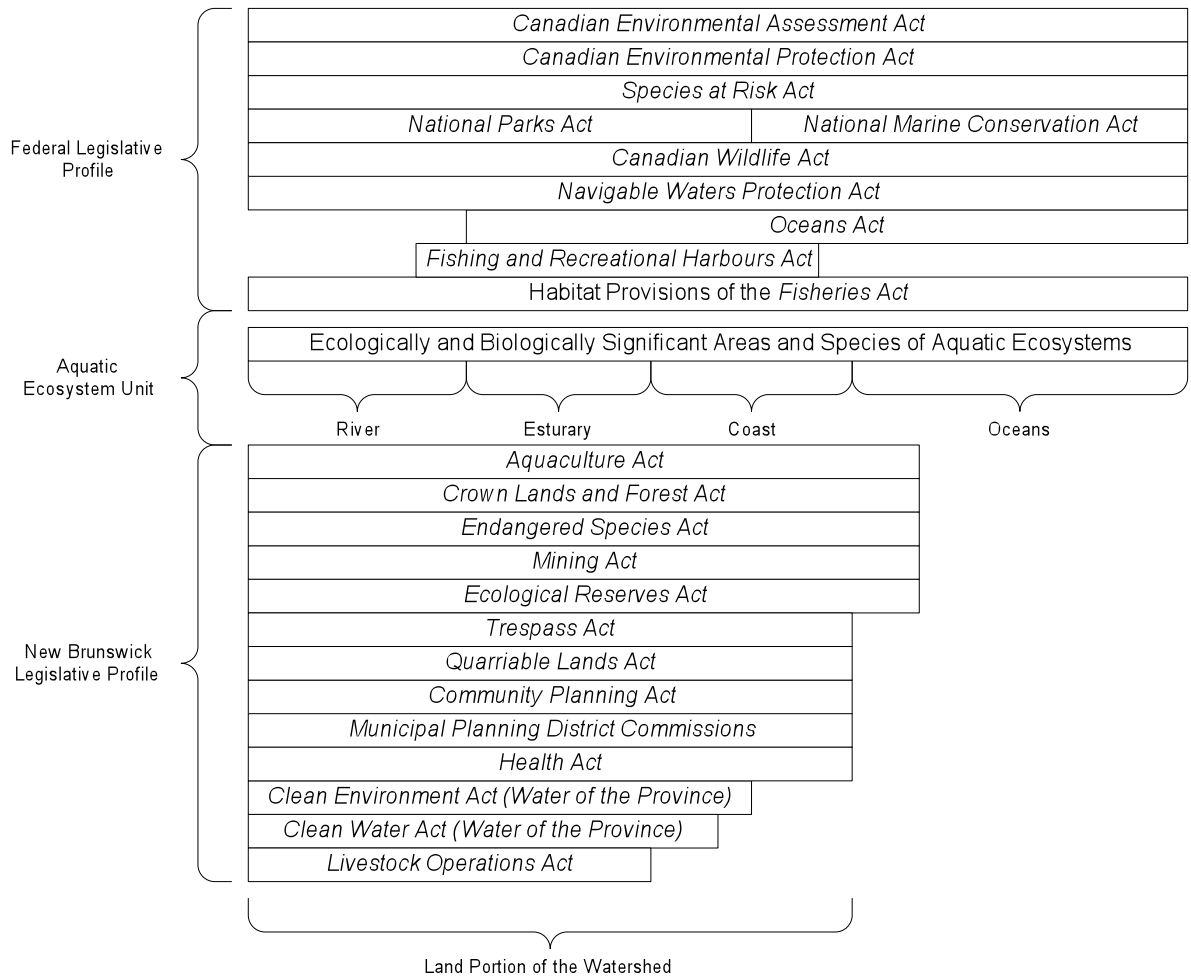


Figure 5. A legislative profile of relevant Canadian federal and provincial acts and regulations that pertain to coastal and marine spatial planning and management along the New Brunswick coastline of the southern Gulf of St Lawrence. Adapted from Cormier, 2010.

3.4. Significant Ecosystem Component Susceptibilities

Eelgrass habitats are spatially restricted to the shallow coastal ecosystems, and so they have become increasingly vulnerable to human activities and coastal developments (Waycott et al., 2009; Schmidt et al., 2011; Schmidt et al., 2012). There is evidence to suggest the declines of eelgrass meadows in the sGSL (Locke, 2005; DFO, 2009; AGRG, 2012) are greater than the global averages¹⁶ with declines of 30-95% reported for some estuaries (DFO, 2009) despite the acknowledgement of their ecological, social, and economic importance (i.e., ESS; DFO, 2009). However, these global declines may be underestimates considering that the research is both incomplete¹⁷ and losses are accelerating¹⁸.

Schmidt and colleagues (2012) have demonstrated that eelgrass habitats in Atlantic Canada are especially vulnerable to various anthropogenic disturbances. Eelgrass declines in the sGSL are mainly attributed to cumulative anthropogenic impacts, namely that of land-based and coastal nutrient loading that cause eutrophication, and in turn increased shading that causes reduced light attenuation for photosynthesis (Lotze et al., 2003; Bastien-Daigle et al., 2007; DFO, 2009; DFO, 2011a; Schmidt et al., 2012). Additionally, eelgrass is negatively affected by climate change and physical damages from coastal development (e.g., dredging) and conversion (e.g., aquaculture) (Thom et al., 2003; DFO, 2009; DFO, 2011a; Schmidt et al., 2012; Skinner et al., 2013).

Oysters have faced a similar fate. Historical oyster reefs were decimated by anthropogenic overexploitation and habitat loss (Lotze et al., 2006), causing an 85% reduction in global reefs in the past 100-150 years (Beck et al., 2011). Comeau and colleagues (2006) had estimated that natural oyster biomass in NB prior to 1990 was approximately 36,000 t but had been reduced to 75 t in 2006; comparatively, it was

¹⁶ The total decline of seagrass habitats has been estimated at 29% worldwide (Waycott et al., 2009) and >65% in selected estuaries and coastal seas (Lotze et al., 2006; Airoidi and Beck 2007).

¹⁷ There is currently a bias within the literature as to which seagrass meadows are sampled (i.e., emphasis on North America, Western Europe, and Australia and deficient data for the North- and South-East Pacifics, Western Pacific, and Southern Atlantic); as a result, the estimates that suggest that seagrasses cover <0.2% of global oceans are highly uncertain (Fourqurean et al., 2012).

¹⁸ The annual rate of global decline has been estimated to be 0.9% prior to 1940, 1.5% from 1940-1990, and 7% from 1990-present (Waycott et al., 2009).

estimated that SBOA had produced approximately 1,250 t of oysters in 2006¹⁹. Based upon these estimates, the combined standing stock (i.e., fishery and aquaculture productions) of oysters in NB estuaries would represent <5% of pre-1990s oyster biomass, which is reportedly consistent with global trends (Bastien-Daigle et al., 2007). In NB, oyster declines were largely attributed to the spread of Malpeque disease from Prince Edward Island in the 1950's, which caused >90% mortality rates; however, it is believed that the surviving oysters were resistant to the disease, to which the existing oyster population are descendants (McGladdery and Bower, 1999). The ongoing loss of natural oyster reefs represents the significant loss to the functional productivity of the ecosystem (Bastien-Daigle et al., 2006).

3.5. Cause-Effect Pathways

Given the number and complexity of the various susceptibilities, pressures, and cause-effect pathways that affect both eelgrass and cultured oysters in the sGSL, only those that relate to the eelgrass-SBOA relationship will be considered henceforth. For a summary of potential impacts onto eelgrass and the ecological repercussions, see Koch (2001) and Cranford and colleagues (2006) (see Appendix 8.3. for pathways-of-effects map for eelgrass). Also, given that societal and economic benefits are closely tied to ecological services in the sGSL (DFO, 2005), the discussion below will largely focus on the ecological components and their effects.

Because both seagrasses and bivalves occupy the same functional niche – low energy sub-tidal marine coastal zones – various species of bivalves live in, on, or near seagrass beds (e.g., Carroll et al., 2008; Wall et al., 2008). Moreover, eelgrass and cultured oysters co-exist along eastern NB, and their distributions often directly overlap in many estuaries (AGRG, 2012; Skinner et al., 2013)²⁰. It has been demonstrated that several interactions and their mechanisms between natural filter-feeder bivalve (e.g., oysters) and marine macrophyte (e.g., eelgrass) communities (McKindsey et al., 2006).

¹⁹ For reasons unknown, the DFO website with 2006 aquaculture statistics (<http://www.dfo-mpo.gc.ca/stats/aqua/aqua06-eng.htm>) have retroactively removed many figures, including those for NB oyster production and value.

²⁰ According to the AGRG (2012) report, the majority of the SBOA was found to exist within eelgrass habitat for all of Richiboucto Estuary, while all the SBOA was found outside of eelgrass habitat in Caraquet. The other examined bays had mixed overlaps.

Under natural conditions, both eelgrass and oyster populations are mutually susceptible to one another's productivity such that the loss of one species can trigger the loss of the other (e.g., Newell and Koch, 2004). It is postulated that historical oyster populations were capable of filtering approximately 80% of Chesapeake Bay, Virginia, compared to the <1% filtration capacity currently observed due to the ongoing loss of natural oyster reefs in the region, in turn having caused a significant loss of water clarification services that led to the decline of eelgrass cover (Newell and Koch, 2004). Inversely, dense eelgrass meadows have been attributed to increase oyster survivability in the face of large storms given that eelgrass can reduce wave energy that would otherwise dislodge bivalves and relocate them to potentially unfavorable habitats (Reusch and Chapman, 1995). These mutual susceptibilities increase the vulnerability of the ecosystem and can cause the significant loss of ecosystem services and functions (Bastien-Daigle et al., 2007). However, the response of eelgrass habitat to natural bivalve reefs differs from that of suspended bivalve aquaculture (e.g., SBOA, McKindsey et al., 2006).

In general bivalve aquaculture can have a broad range of effects onto coastal ecosystems and at different scales (Bastien-Daigle et al., 2007). In fact, the influences of suspended bivalve aquaculture can have opposing effects onto macrophyte habitats (Newell, 2004; Anderson et al., 2006; McKindsey et al., 2006); that is, the effects of SBOA onto eelgrass can be positive, negative (Dumbauld et al., 2009; Tallis et al., 2009), or inconsequential. The extent of these effects are contextualized at the specific study sites and their local environments, with the variability and vulnerability dependent upon various spatial and temporal factors (e.g., hydrology and the rate of tidal mixing, SBOA population density, phytoplankton concentrations and primary productivity) (Newell, 2004; Anderson et al., 2006; Cranford et al., 2006; McKindsey et al., 2006; Bastien-Daigle et al., 2007). As a result, these potentially opposing effects are complex, non-linear (Bastien-Daigle et al., 2007), and can proliferate at the local (i.e., near-field and zone of influence) to system (i.e., bay, estuary scales) to SES scales (Dumbauld et al., 2009). Therefore, comprehensive studies to determine whether SBOA has a net positive or negative effect on coastal ecosystems have thus far been limited (Anderson et al., 2006, McKindsey et al., 2006).

The context (e.g., scope, management objective) of the eelgrass-SBOA relationship must be adequately defined in order to assess the net effects at each spatial scale and inform the risk assessments²¹. At the local scales (i.e., near-field and ZOI scales), the suggested management objective is to protect local eelgrass patches and conserve the community structures and functions that they support (e.g., nursery, foraging, feeding habitats for fish and invertebrates). Of the various identified sources of HADD onto eelgrass, SBOA is a primary consideration given the direct or adjacent overlap in distributions; the direct and indirect, positive and negative effects of SBOA onto eelgrass at the local scale are outlined below (see sections 3.5.1. and 3.5.2. below). At the larger ecosystem scales (i.e., bay and estuary scales), the suggested management objective is to protect the entire eelgrass meadow as it is distributed throughout the ecosystem and to conserve the services and functions the eelgrass provides to the coastal system (e.g., lessen turbidity, reduce wave attenuation and energy, protect coastal lands, provision dissolved oxygen). At the ecosystem scale, the adverse negative effects that cause HADD onto eelgrass are expanded to include nutrient and sediment loading (although this also affects eelgrass at the local scale, these effects are now enhanced due to scaling; pers. comm., Marc Ouellette, August 20, 2013), and so anthropogenic pressures that are not considered at the local scale, such as land-based agriculture and infrastructure, are now major concerns. Therefore, watershed land-uses are considered at the ecosystem scale. However, the land-based effects were expected to be most pronounced at the estuary rather than bay scale given the difficulty in distinguishing the distribution of run-off into each bay within a watershed and because of the unique hydrological and oceanographic processes that dissimilarly assimilate the nutrient and sediment loading. The direct and indirect, positive and negative effects of SBOA onto eelgrass at the ecosystem scale are outlined below (see sections 3.5.3. and 3.5.4. below).

For the purpose of this research project, it has been attempted to limit the following subsections to the positive and negative effects of increased²² SBOA gear and

²¹ Although it is most recommended to use the DSPIR (Drivers–Pressures–State Change–Impact–Response) framework (pers. comm., Marc Ouellette, August 20, 2013; refer to Atkins et al., 2011), the limitations of this project will only allow for an abbreviated description.

²² Given that the SBOA industry is expected to further increase production in the near future, in turn increasing the gear used and the filtration capacity of SBOA.

filtration onto eelgrass at multiple spatial scales in eastern NB and the sGSL. Although largely discussed throughout the literature, the effects of biodeposition and benthic nutrient loading will be omitted here given that there is adequate tidal mixing along eastern NB and the sGSL to prevent negative habitat effects²³ (Mallet et al., 2006; Bastien-Daigle et al., 2007).

3.5.1. Near-field Scale SBOA Effects

In accordance with Skinner and colleagues' (2013) results, the near-field scale has been identified as the area directly under and between SBOA gears with a maximum 25 m radius from the SBOA site. Empirical evidence has suggested that the greatest negative effects of SBOA onto eelgrass are caused by the structures and gears (e.g., floating bags, cages, long-lines) that directly cause shading, smothering, and physical damages year-round. Skinner and colleagues (2013) have demonstrated that shading is the primary negative impact given that it causes significant reductions in eelgrass distribution, productivity, and photosynthetic rates so as to decrease eelgrass biomass by as much as 79% within 25m of the SBOA site. It was found that SBOA stocking densities and the age of the SBOA lease were the primary variables responsible for the damages caused by shading, and that stocking density explained 35% of the variation in eelgrass cover in the three examined eastern NB estuaries (Skinner et al., 2013). Second, the over-wintering of SBOA gear onto eelgrass meadows has been demonstrated to cause long-term damages; suspended long-line structures reduced the structure and productivity²⁴ of eelgrass after just 67 days with incomplete recovery after 645 days (Skinner et al., submitted). If the examined eelgrass fails to recover, these over-wintering effects could qualify as permanent damages (no recover after >5 years) supporting fish habitat and would therefore qualify as a violation under the new

²³ Bastien-Daigle and colleagues (2007, p. 40) write that, "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."

²⁴ Structure was measured by shoot density, biomass, canopy height; productivity was measured by growth rate, photosynthetic rate/efficiency (Skinner et al., 2013; Skinner et al., submitted).

Fisheries Act (Koops et al., 2013). Additionally, eelgrass is negatively impacted from the SBOA harvesting process, in which incidents of mooring, boat wash, and trampling can physical damage the meadows (Bastien-Daigle et al., 2007).

A fourth pathway of effect at the near-field scale can result from the SBOA structures offering an addition of substrate, which can have negative impacts onto eelgrass but positive or negative ecosystem effects. The addition of substrate can increase habitat complexity within the water column to support a greater abundance and diversity of organisms and trophic levels (McKindsey et al., 2006; Bastien-daigle et al., 2007) to improve ecological health. However, the addition of some species can increase competition for light, nutrients, and carbon, and therefore become detrimental to eelgrass productivity and therefore degrade ecosystem health (Koch, 2001; Dumbauld et al., 2009). Ultimately, the effects of SBOA at the near-field scale have been widely demonstrated to negatively impact eelgrass meadows in eastern NB by reducing productivity, reproductive viability, and survival (Skinner et al., 2013).

3.5.2. Zone of Influence Scale SBOA Effects

The ASC (2012) defines the ZOI as the area of phytoplankton depletion caused by bivalve aquaculture where enhanced filtration is the dominant factor at the ZOI scale. Oyster filtration is an essential coastal ecosystem service since it removes phytoplankton and therefore improves water clarity, in turn increasing light attenuation for enhanced eelgrass photosynthesis and productivity (e.g., Newell & Koch 2004; Bastien-Daiglet et al., 2007; Wall et al., 2008; Forrest et al., 2009; Tallis et al., 2009; Comeau, 2013). It is widely suggested that this filtration service is the primary benefit of bivalve aquaculture (Tallis et al., 2009; Comeau, 2013). As a result of oysters filtering phytoplankton from the water column, bivalves have the capacity to exert a top-down control on phytoplankton communities (Newell and Koch, 2004; Wall et al., 2008; Dumbaul et al., 2009), although recent research by Comeau (2013) suggests that the oysters' current capacity in eastern NB to exert top-down control is on the order of 10-14 times less in comparison to that of historical oyster populations. However, top-down control by oysters on phytoplankton may be limited by their simultaneous bottom-up control. Oysters are highly selective as to the plankton they consume (Newell and Koch, 2004), and the removal of algal cells may promote primary production during periods when

phytoplankton would otherwise be limited (Prins et al., 1997; Anderson et al., 2006). Ultimately, this could enhance eutrophication, potentially beyond the control of bivalve filtration (McKindsey et al., 2006).

3.5.3. Bay Scale SBOA Effects

The bay scale is simply defined by the area of the bay, as established by the Government of Canada with geographical boundaries and area data supplied by DFO. The far-field effects of SBOA onto eelgrass meadows have not yet been well described for eastern NB and the sGSL (Anderson et al., 2006). Recently, Wagner and colleagues (2012) analyzed the bay-scale effects onto eelgrass in Willapa Bay, Washington, caused by different cultured oyster stocking densities, using the oysters' shells and biodeposition as co-variables. It was found that the oysters were spatially competing with eelgrass and could cause significant declines when oyster densities exceeded 20% cover of bay area (Wagner et al., 2012). Comparatively, the bays examined for this project have a mean 8.1% cultured oyster cover (percent mean TAL per mean bay area), perhaps suggesting that eastern NB eelgrass is not yet adversely affected by oyster aquaculture given that it has not yet reached the critical threshold suggested by Wagner and colleagues (2012).

Similarly, Newell and Koch (2004) had modeled the bay-scale filtration effects of cultured oysters at modest and high abundances²⁵ and found that they had promoted eelgrass growth and increased shoot densities, respectively, due to the ability of SBOA gear to reduce wave energy and height along Maryland. These results suggest that relatively low densities of cultured oysters (25 g/m²) are sufficient to perform critical ecological functions that promote eelgrass productivity. However, oyster densities along eastern NB were found to be far less (approximately 4 g/m²; Locke, unpublished data) than the stocking densities tested by Newell and Koch (2004). It is uncertain if such low SBOA stocking densities along eastern NB could similarly diffuse wave energy like those predicted Maryland.

²⁵ Modest and high oyster densities were modeled to be 25 and 75 g/m² (aggregate dry tissue weight [ADTW]), respectively. It should also be noted that the model had assumed uniform distribution of oysters throughout the domain, which is not realistic for SBOA. Locke (unpublished data) had estimated a mean of approximately 4 g/m² for eastern NB SBOA.

Based upon the manipulated SDI model created by Guyondet and colleagues (2013), the cultured oysters in Aldouane and Bedec appear to have significant water filtration rates²⁶ sufficient to deplete phytoplankton biomass throughout the observed bays and beyond (see Appendix 8.4.1 for detailed results). Although the SBOA stocking densities in eastern NB are considerably lower than regional comparisons (e.g., Newell and Koch, 2004; Wagner et al., 2012), numerical models for local bays suggest that SBOA has the propensity to exert a significant effect on bay-scale primary productivity and therefore food webs. It is possible that sustained seston depletion could alter the long-term species composition of phytoplankton communities and potentially enhance bay-scale eutrophication in bays with slow water renewal time (e.g., Bedec) and would ultimately become detrimental to eelgrass meadows (Anderson et al., 2006; McKindsey et al., 2006). Moreover, the sustained phytoplankton depletion in the Richiboucto would be indicative that SBOA is beyond the natural carrying capacity limits of these bays.

3.5.4. Estuary Scale SBOA Effects

Similar to the bay-scale, the estuary-scale is simply defined by the area of the estuary, as established by the Government of Canada and geographical boundaries and area data supplied by DFO. For the purpose of this project, an estuary can be comprised of one or more bays (e.g., Miscou is included as both a bay and an estuary; Tracadie estuary is comprised of 2 bays: Tracadie South and North). At the estuary-scale, eelgrass meadows become vulnerable to multiple anthropogenic pressures given the influence of surrounding land-based uses and activities within the estuaries' watersheds. Among these pressures, eutrophication continues to be among the greatest contemporary threats onto eelgrass and seagrasses worldwide (Waycott et al., 2009; Schmidt et al., 2012). The association of land-use to coastal ecosystem health is largely recognized, and the agriculture industry has been largely attributed to ecosystem health declines (e.g., Cairns, 2002; Cranford et al., 2006). Anthropogenic eutrophication is largely caused by nutrient-loading from land-based sources that enrich nutrient concentrations in the water column; although the effects of nutrient enrichment are highly contextualized to the specific bay and estuary, the application of agricultural fertilizers is renowned to cause

²⁶ Oyster in Richiboucto have been estimated to clear the water at a rate of 2.57L oyster⁻¹ hour⁻¹ (Guyondet et al., 2013).

eutrophication (e.g., Meeuwig, 1999). Land-sourced eutrophication can have both direct and indirect negative effects onto eelgrass meadows (*see* Cloern, 2001, for greater descriptions; DFO, 2011a; Schmidt et al., 2012) as well as bivalve aquaculture (e.g., Meeuwig, 1999). Research by Schmidt and colleagues (2012) demonstrated that under highly eutrophic conditions, eelgrass meadows along eastern NB and PEI are susceptible to an ecological shift towards more opportunistic algae populations, therefore catalyzing the loss of their critical structures and services that can have negative secondary consequences onto the SES.

Given that oysters have relatively high filtration rates and can significantly deplete estuaries of phytoplankton (Guyondet et al., 2013), SBOA has been promoted as a mechanism to mitigate the effects of nutrient enrichment and coastal eutrophication (Cranford et al., 2003; Newell, 2004; Anderson et al., 2006). As a regional comparison, examples of intense mussel aquaculture practices (i.e., high stocking densities) in PEI estuaries were found to overcompensate the effects of nutrient loading by increasing filtration²⁷ and therefore phytoplankton depletion rates (Landry, 2002; Anderson et al., 2006). Moreover, Guyondet and colleagues (2012) demonstrated that the cultured oysters' filtration capacity in Richiboucto estuary was sufficient to deplete phytoplankton from adjacent bays from which the SBOA sites were localized (i.e., the SBOA sites in Bedec caused a reduction in seston concentration in the adjacent Richiboucto Harbour). These results suggest that the estuary-scale SBOA effect onto phytoplankton is significant, and therefore has the propensity to exert a top-down control on phytoplankton that could benefit of eelgrass via improved light attenuation (Anderson et al., 2006). Additionally, oysters have sufficient phenotypic plasticity to increase filtration rates in response to increased phytoplankton concentrations (Landry, 2002, in Cairns et al 2002), suggesting that they can mitigate the negative effects of increasing land-based nutrient loading from the surrounding watersheds (e.g., Anderson et al., 2006) and improve coastal conditions for eelgrass productivity.

However, research by Meeuwig (1999) suggested that mussel aquaculture production rates had become dependent upon the high phytoplankton primary

²⁷ It is important to note that the filtration rate of mussels is approximately 3-times less than that of oysters (Comeau, 2006)

production rates supported by the large agricultural nutrient loading inputs. Ultimately, the research had suggested that the cultured mussels had exerted a significant bottom-up control on phytoplankton communities so as to concentrate eutrophication impacts (in turn increasing pressures onto eelgrass) and increase cultured mussel mortality rates²⁸ (Anderson et al., 2006). It is possible that a similar situation could arise along eastern NB with cultured oysters; the SDI numerical model (Guyondet et al., 2013) had predicted that the estuary-scale filtration effects were sufficient to significantly deplete phytoplankton in adjacent bays where SBOA was modeled to be absent (see Appendix 10.4.1 for detailed results). Moreover, results showed that the cultured oysters were removing phytoplankton faster than its regeneration time²⁹, suggesting that similar shifts in seston community structures could cause negative ecosystem effects (ASC, 2012), including eutrophication. Comparatively, agriculture land-use in eastern NB watersheds is significantly less than those in PEI³⁰ and so similar nutrient loading impacts are not expected in NB estuaries. Ultimately, the cultured oysters' top-down control on phytoplankton communities and ability to mitigate the effects eutrophication may not be sustainable in the long-term given the simultaneous bottom-up control (e.g., Asmus and Asmus, 1991; McKindsey et al., 2006). Therefore, while watershed development and its land-based nutrient run-off may be mitigated by SBOA in the short-term, it cannot be guaranteed that these mitigation effects can be sustained in the long term.

²⁸ By having selectively grazed upon small phytoplankton species, the mussels had shifted the seston community structure to consist of only the remaining larger, non-edible phytoplankton species. Over time, these remaining phytoplankton communities had become so abundant as to cause eutrophic conditions. Simultaneously, the loss of the smaller, desirable phytoplankton species had caused a shortage of available food for the mussels, therefore increasing intraspecific competition and mortality rates (Anderson et al., 2006).

²⁹ According to the ASC (2012), primary production time (PPT) should be less than oyster clearance time (CT) (i.e., $CT/PPT > 1$) in order to avoid negative ecosystem effects from phytoplankton depletion; when $CT/PPT \leq 3$ then the aquaculture site is no longer viable for certification and must be subjected to a bay-wide management plan (ASC 2012). The SDI numerical model created by Guyondet and colleagues (2013) found that $PPT/CT = 3$ for the Richiboucto estuary, which when converted to the ASC standards, is $CT/PPT = 0.33$ and is therefore in excess of local ecological carrying capacity. Under the ASC (2012) recommendations, the Richiboucto estuary should therefore be subjected to a bay-scale management plan – one that individually considers each of the three contributory bays – in order to ensure that the SBOA is not outstripping the capacity of the ecosystem to replenish phytoplankton.

³⁰ The mean agriculture land-use area per watershed in NB is 7.4%; in PEI, the mean is 44% (Finley et al., 2013).

4. Risk Assessment Results

4.1. Near-Field Scale Risk Assessment

Research from Skinner and colleagues (2013) and the preliminary overwintering study (Skinner et al., submitted) have been given the greatest amount of influence for the near-field risk assessment. The likelihood of SBOA gear to cause damages to underlying and adjacent eelgrass beds has been classified as “likely” (76-95% occurrence) in accordance with Skinner and colleagues’ (2013) given that not all sample sites were found to elicit a negative response (pers. comm. Marc Skinner, August 15, 2013). The preliminary overwintering results suggest similar occurrence of damages (pers. comm. Monica Boudreau, May 1, 2013).

The ecological impacts onto eelgrass at the near-field scale have been categorized as a “very high” risk. Recent research has demonstrated that the effects of shading caused by SBOA gear is a significant source of negative impacts onto eelgrass causing the reduced capacity and rate of eelgrass photosynthesis by a factor of 10, in turn limiting the productivity, viability, and survival of eelgrass (Skinner et al., 2013). Using the active SBOA lease (ASL) area as a proxy for similar near-field damages, it is estimated that a mean of 45.8 ha of eelgrass, or mean 2.3% eelgrass per bay area, is already at risk in the n=14 examined bays along eastern NB. It is important to note, however, that Skinner and colleagues (2013) observed near-field effects up to 25 m from the SBOA site, and so the above ASL-based estimate is likely an under-estimate. Moreover, the smothering effects of overwintering SBOA gears have shown a minimum of three years of permanent damages onto underlying eelgrass meadows (Skinner et al., submitted). As a result, the loss of eelgrass can alter the ecological structure and function at the localized near-field scale, in turn reducing the availability of eelgrass as nursery and other habitats to fish and invertebrate species. This could further allow for the introduction of more competitive and invasive algal species, like *Ulva lactuca*, causing the endangerment of local eelgrass meadows and potentially catalyzing their extirpation in some areas.

The socio-economic consequences are ranked as a “low” risk. Existing SBOA practices have been proven to cause severe and potentially long-term damages to eelgrass (Skinner et al., submitted) and therefore reduces the amount of critical habitat

available to fish, in turn reducing the access to the socio-economically important CRA fisheries. However, these eelgrass impacts are localized, and would not necessarily cause the significant reduction to access to the CRA fisheries on the larger scale. Moreover, because the SBOA industry is not currently responsible for monitoring and mitigating their effects onto eelgrass due to the lack of mandated regulations, the near-field effects are not directly affecting the social and economic sectors. The SBOA industry will only accrue routine costs in order to comply with RCSR (Transport Canada, 2007) requirements. Contrastingly, if the regulations were sufficient to mandate such monitoring and mitigation programs, the socio-economic sector would have to be aware of these effects and address them, costing the SBOA industry both time and money.

Likewise, the operational repercussions of the near-field effects were ranked as “medium-low” risks. The minimal legal requirements for the SBOA industry to adjust their practices and gears to reduce harm onto eelgrass have allowed them to avoid costly modifications, despite the possible eelgrass benefits. Interestingly, the SBOA industry has been recently trending towards an increase in long-line systems, allowing the aquaculturalists to yield a greater market price in less time for their cultured oysters; in turn, the less dense stocking densities benefit local eelgrass meadows due to reduced shading and increased light attenuation (pers. comm., Marc Skinner, June 12, 2013). Should the SBOA industry become mandated to adopt long-line practices, it can be expected that there would be minor to moderate disruptions to their operation while they reallocate resources, potentially resulting in the loss of assets or over-spending of their budgets by \$100,000- 1 million.

The strategic repercussions were ranked as a “medium” risk given the considerable lack of public and private interest or awareness in the local condition of eelgrass meadows. However, because eelgrass as critical fish habitat is directly linked to the CRA fisheries (DFO, 2013a) and was until recently protected by federal law (i.e., Fisheries Act, 1985), the decline of eelgrass at the near-field scale is representative of the government’s failure to conserve eelgrass and realize its mandates. As a result of the local loss of eelgrass, the needs of those whose livelihoods are dependent upon the near-shore fisheries will go unmet as the habitat for their CRA fisheries decline. This could in

turn cause the mistrust of these stakeholders as they recognize that the SBOA industry and its effects onto eelgrass obstruct their localized access to the fisheries and their livelihoods. The near-field results are summarized below.

Table 4. Risk analysis of the near-field effects of SBOA onto underlying eelgrass meadows. The likelihood has been ranked as a 4 (likely) dependent on the fact that 76-95% of case studies demonstrated significant changes between/under leases compared to reference sites. The overall impacts and consequences for near-field effects have been ranked as a 3 (medium; rounded from 2.85) upon averaging the individual rankings for the ecological, socio-economic, operations, and strategic consequences.

Likelihood: 4. Likely			
Risk Analysis of Impacts and Consequences		Risk Analysis of the Repercussions Tolerability	
Impacts: Ecological 4. Very High Risk	Consequences: Socio-Economic 2. Low Risk	Repercussions: Operational 2.5. Medium-Low Risk	Repercussions: Strategic 3. Medium Risk
<ul style="list-style-type: none"> ➤ Major shift in species composition (e.g., introduction of invasive species) ➤ A species is listed as endangered or locally extinct; recovery plan required ➤ Significant damage to critical fish habitat with limited ability to recover 	<ul style="list-style-type: none"> ➤ Reduced access to commercial, recreational, and aboriginal (CRA) fishing activity ➤ Routine cost (mitigation measures) to developer to accommodate habitat requirements 	<ul style="list-style-type: none"> ➤ [Limited and temporary] disruption of operations, requiring minor to moderate reallocation of resources within the area, regionally or outside of programs ➤ Overspending of the budget between \$100,000-1 million CAD ➤ Loss of assets by \$100,000-1 million CAD 	<ul style="list-style-type: none"> ➤ Some loss of client trust, escalating resistance ➤ Some client and/or public needs go unmet ➤ Sustained negative international media attention/public criticism ➤ Erosion of DFO and/or Canada’s international reputation ➤ Failure to achieve a key departmental or government objective (Very High Risk)

The overall near-field scale risks of SBOA onto eelgrass was ranked as a “medium” upon assigning each of the four categories from 1 to 5 and finding their average. Plotted against the likelihood of the risks, which was ranked as “likely”, the resulting heat analysis map reveals that current conditions are not tolerable (see Fig. 6 below). In accordance with the IOM Risk Management Handbook (see Cormier et al., 2013), these results are indicative that effective ICZM measures are required to mitigate the effects at the near-field scale; in particular, the ecological impacts must be immediately mitigated with management.

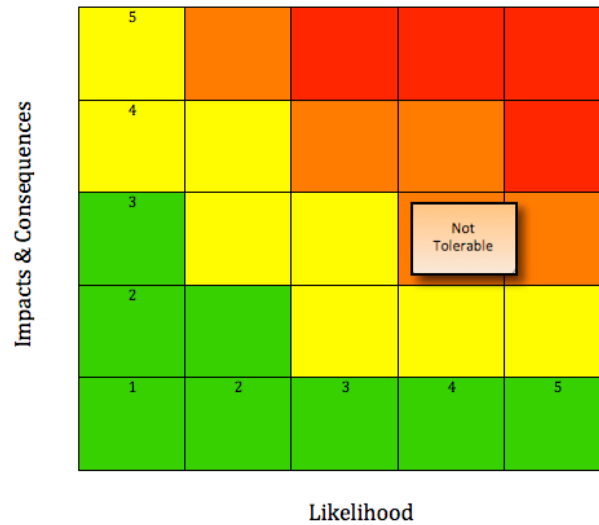


Figure 6. Risk analysis heat map for the near-field effects of SBOA onto eelgrass. Given that the risk analysis results have demonstrated a “likely” likelihood, and that the overall impacts and consequences averaged to be “medium”, the near-field eelgrass-SBOA relationship exudes “not tolerable” conditions and thus demands ecosystem-based risk management and treatment options.

4.2. Zone of Influence Scale Risk Assessment

The predictive results from the Guyondet and colleagues’ (2013) numerical model have been given the greatest amount of influence for the ZOI risk assessment. The likelihood of phytoplankton depletion in conjunction with SBOA sites is predicted to be an almost certain (>95% likelihood) risk, although these effects are contextualized to the Richiboucto estuary and may not be as likely for other estuaries. Therefore, the likelihood of the ZOI effects on eelgrass was ranked as a “likely” (76-95% likelihood) risk to reflect the fact that other bays and estuaries are not expected to behave like the Richiboucto unless they shared similar hydrodynamic and oceanographic conditions.

The ecological impacts of phytoplankton depletion within the SBOA sites’ ZOI are not well understood. Certainly, the ZOI has significant and far-reaching effects onto seston concentrations in the respective bay as well as adjacent bays (see Appendix 10.4.1. for ZOI results), potentially allowing for greater eelgrass productivity as a result of better light attenuation as well as eutrophication mitigation. However, research remains to explore the long-term effects of phytoplankton depletion from SBOA filtration given that oysters are selective as to the size of plankton they consume, which may cause an ecological shift in phytoplankton communities towards smaller species and enhance long-term effects of eutrophication. However, phytoplankton depletion is a

seasonal occurrence and seston is known to accumulate with every spring bloom. Therefore, it may or may not be beneficial to eelgrass to have large and overlapping ZOI like those observed in Richiboucto. Regardless, the ecological impact can be ranked as “low” risk given the apparent benefit to eelgrass and the annual recovery of the phytoplankton communities.

The socio-economic consequences of SBOA onto eelgrass at the ZOI scale were ranked as a “negligible” risk. Although high seston depletions within the ZOI will likely reduce phytoplankton availability to other filter feeders (e.g., wild oyster), the reality is that there are few remaining fishers whose livelihoods depend on wild bivalves. Additionally, with the potential for eelgrass to benefit from the ZOI and because eelgrass is a critical habitat for many CRA fisheries, there is a propensity for minor increases in access to the CRA fisheries. Moreover, the lack of legislation and regulations to monitor eelgrass, little alone phytoplankton concentrations, mean that the aquaculturalists and other fishers will accrue minor to no financial costs to accommodate eelgrass’ habitat requirements (e.g., high light attenuation, eutrophication mitigation). Likewise, these minor costs translate into a “negligible” risk for operational repercussions such that there is limited to no disruption of the current SBOA practices and therefore no need to reallocate resources. Lastly, given that the government could positively interpret these results and would disseminate an optimistic perspective to the public, it is likely that there are “negligible” risks associated with the strategic repercussions of SBOA services within the ZOI scale.

Table 5. Risk analysis of the zone of influence (ZOI) effects of SBOA onto surrounding eelgrass meadows. The likelihood has been ranked as a 4 (likely) given that it is assumed that 76-95% of case studies would demonstrate similar ZOI effects. The overall impacts and consequences for ZOI-scale effects have been ranked as a 1 (negligible; rounded from 1.25) upon averaging the individual rankings for the ecological, socio-economic, operations, and strategic consequences.

Likelihood: 4. Likely			
Risk Analysis of Impacts and Consequences		Risk Analysis of the Repercussions Tolerability	
Impacts: Ecological <i>2. Low Risk</i>	Consequences: Socio-Economic <i>1. Negligible Risk</i>	Repercussions: Operational <i>1. Negligible Risk</i>	Repercussions: Strategic <i>1. Negligible Risk</i>
➤ Minor, recoverable short-term (1 year), e.g., seasonal changes in habitat and phytoplankton communities	➤ Minor reductions in access to commercial, recreational, and aboriginal (CRA) fishing activity ➤ Minor cost to the developer to accommodate habitat requirements	➤ Minor to no disruption of program operations, with no reallocation of resources	➤ Minor but recoverable set-back in client/public trust

The overall ZOI-scale risks of SBOA onto eelgrass were ranked as a “negligible” risk upon assigning each of the four categories from 1 to 5 and finding their average. Plotted against the likelihood of the risks, which was ranked as “likely”, the resulting heat analysis map reveals that current conditions are extremely tolerable (Figure 7). According to the IOM Risk management guidebook, immediate and effective management is not needed to mitigate the effects of SBOA onto eelgrass at the ZOI scale.

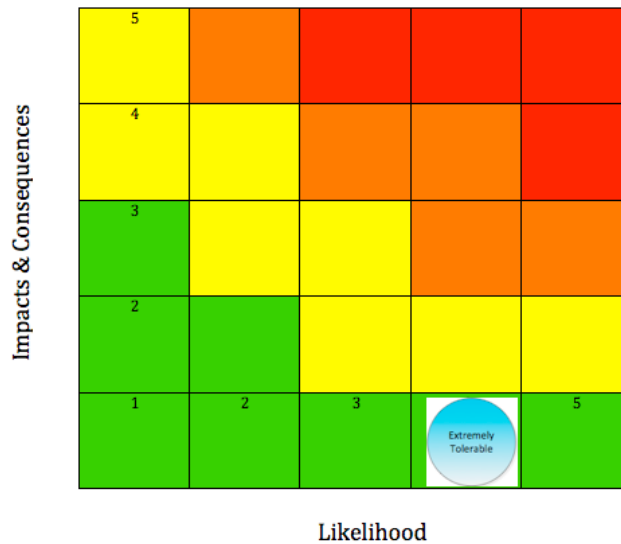


Figure 7. Risk Analysis Heat Map for the ZOI effects of SBOA onto eelgrass. Given that the risk analysis results have demonstrated a “likely” likelihood, and that the overall impacts and consequences averaged to be “negligible”, the ZOI-scale eelgrass-SBOA relationship exudes “extremely tolerable” conditions and does not demand immediate ecosystem-based risk management and treatment options.

4.3. Bay-Scale Risk Assessment

The results from the bay-scale statistical analyses (see Appendix 8.4.2. for all results) as well as the available literature were used to inform the following risk assessment. It has been demonstrated from the MRA results that there is “almost certain” likelihood (>95 likelihood, or $P < 0.05$) of each of the examined SBOA co-variables to significantly affect eelgrass cover (ha), whether positively or negatively, with the results indicating highly correlated but contextualized bay-scale SBOA effects ($P < 2e-16$ for $n=10/11$ co-variables; $R^2 = 0.99$)³¹. The literature additionally supports a generally high correlation between eelgrass health and filter feeders (e.g., Tallis et al.,

³¹ Although the MRA had revealed that all of the co-variables considered have a significant influence onto eelgrass cover at the bay-scale, it is not certain that all of the possible co-variables that could affect eelgrass cover have been included in the model.

2009), and given that local wild suspension-feeding oyster populations have been sharply reduced (e.g., Bastien-Daigle et al., 2007), the predominant bivalve influence is derived from the aquaculture industry and specifically SBOA. Therefore, there is confidence in ranking the likelihood of the bay-scale eelgrass-SBOA relationship and effect as an “almost certain” risk.

The ecological impact of SBOA on eelgrass was analyzed using the statistical analyses (particularly the SA) as well as the literature, and it was determined that the bay-scale effect was a “medium” risk. The SA results suggested that the SBOA effects onto eelgrass are contextualized to both the bay and SBOA characteristic examined. That is, while some bays have the propensity to have improved conditions and eelgrass coverage due to the increase in SBOA, other bays are predicted to experience a detrimental ecological effect from increased SBOA. For the latter, the risk to eelgrass at the bay-scale has the propensity to be significant given that some bays are predicted to have eelgrass cover declines, potentially causing long-term damages. However, appropriate management measures (i.e., EBRM) can mitigate these risks to minimize ecological harm and possibly improve eelgrass and ecosystem health.

The bay-scale socio-economic consequences were ranked as a “low” risk to reflect the contextualized eelgrass response to different SBOA conditions. Eelgrass is a critical habitat for many economically viable fisheries, and so the potential bay-scale loss of eelgrass could reduce access to the CRA fisheries while an increase in eelgrass could increase access. It is important to note that there is a greater economic incentive for the SBOA industry to produce oysters at a level beyond the bay’s ECC, whereas there is no direct economic cost for exceeding the eelgrass’ ECC. However, given the current lack of strong legislation to regulate the SBOA industry and its effects at the bay-scale, aquaculturalists are not mandated to accommodate the eelgrass’ ECC, and are therefore not required to employ monitoring measures or incur mitigation costs. Therefore, the majority of the SBOA industry will continue to incur only routine costs.

Where some bays will benefit from additional SBOA activity while others will not, the operational repercussions were ranked as a “low” risk to represent the fact that some aquaculturalists will experience some disruptions while others will not. Again, the lack of legislation allows for the SBOA industry to independently allocate its resources,

and neutral and positive eelgrass responses to changes in SBOA characteristics will not cause any disruptions to the aquaculturalists and their practices. However, in the event of a negative eelgrass response and resulting bay-scale impacts (e.g., increased sedimentation), the aquaculturalist may experience a financial cost due to the loss of assets (e.g., oyster die-off). It is unclear how much financial loss could be attributed to eelgrass cover declines, although costs are only expected to result from significant negative declines, which are only expected for a few bays under specific SBOA conditions. Financial costs can be proactively minimized with small changes to operations to increase monitoring and mitigation measures to ensure the prevention of eelgrass loss, which would in turn cause an overspending of in SBOA budget.

The strategic repercussions were ranked as a “medium-low” risk given that there is a contextualized risk onto eelgrass. In cases where bay-scale eelgrass cover was predicted to decline in response to greater SBOA activity, there is a possibility of lost public trust and cooperation, perhaps due to the exclusion from critical stakeholder meetings or the loss of access to the CRA fisheries. In the most extreme cases there could be the expected negative media attention and international reputation for which DFO/ Government of Canada would be responsible to manage. However, given that some bays experience a positive interaction with increased SBOA, it is expected that at most there would be a loss of trust and escalating resistance for aquaculturalists to adhere to DFO mandates. The bay-scale risk assessment results are summarized below.

Table 6. Risk analysis of the bay-scale effects of SBOA onto eelgrass cover. The statistical analyses have revealed a significant probability between SBOA characteristics and eelgrass cover ($P=2.16e-16$), and so the likelihood of bay-scale SBOA effects was ranked as a 5 (almost certain; >95% likelihood). The overall impacts and consequences for bay-scale effects have been ranked as a 2 (low; rounded from 2.38) upon averaging the individual rankings for the ecological, socio-economic, operations, and strategic consequences.

Likelihood: 5. Almost Certain			
Risk Analysis of Impacts and Consequences		Risk Analysis of the Repercussions Tolerability	
Impacts: Ecological <i>3. Medium</i>	Consequences: Socio-Economic <i>2. Low Risk</i>	Repercussions: Operational <i>2. Low Risk</i>	Repercussions: Strategic <i>2.5. Medium-Low Risk</i>
<ul style="list-style-type: none"> ➤ Species is under significant pressure or at historic lows ➤ Moderate impact to fish habitat, with longer term (i.e, 3 to 5 years) for recovery 	<ul style="list-style-type: none"> ➤ Reduced access to commercial, recreational, and aboriginal (CRA) fishing activity ➤ Routine cost (mitigation measures) to developer to accommodate habitat requirements 	<ul style="list-style-type: none"> ➤ Limited and temporary disruption of operations, requiring minor to moderate reallocation of resources within the area, regionally or outside of programs ➤ Overspending of the budget between \$100,000 ➤ Loss of assets by \$100,000 	<ul style="list-style-type: none"> ➤ Some loss of client trust of cooperation, escalating resistance ➤ Some client/public needs go unmet ➤ Sustained negative national media attention / public criticism ➤ Erosion of DFO / Canada’s international reputation

The overall bay-scale risks of SBOA onto eelgrass were ranked as a “low” upon assigning each of the four categories from 1 to 5 and finding their average. Plotted against the likelihood of the risks, which was ranked as “almost certain”, the resulting heat analysis map reveals that current conditions are tolerable (Fig. 8) and do not demand immediate ecosystem-based risk management and treatment options.

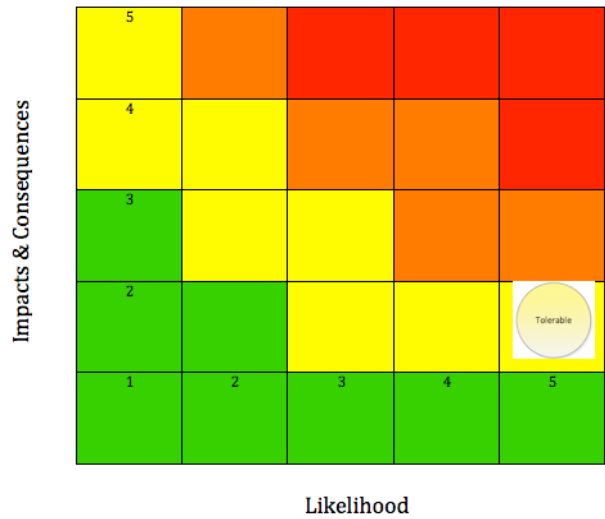


Figure 8. Risk Analysis Heat Map for the bay-scale effects of SBOA onto eelgrass. Given that the risk analysis results have demonstrated an “almost certain” likelihood, and that the overall impacts and consequences averaged to be “low”, the near-field eelgrass-SBOA relationship exudes “tolerable” conditions and does not demand immediate ecosystem-based risk management and treatment options.

4.4. ESTUARY-SCALE RISK ASSESSMENT

The results from the estuary-scale statistical analyses as well as the available literature were used to inform the following risk assessment. It has been demonstrated from the MRA results that there is a significant probability of each of the SBOA co-variables examined to effect and correlate to eelgrass cover (ha) ($P = <2e-16$ for all co-variables; $R^2=0.89$), both positively and negatively, with the results indicating significant estuary-scale SBOA effects. However, the effect of SBOA onto eelgrass is less intensive at the estuary-scale than at the bay-scale given that there are several additional sources of pressure, including watershed-scale land-uses and run-off. Because the statistical analyses could not simultaneously consider watershed and estuary-scale co-variables, the likelihood of eelgrass being affected by SBOA at the estuary-scale was given a conservative estimate of “likely” (76-95% likelihood), representing both positive and negative risks.

The ecological risks of SBOA onto eelgrass at the estuary-scale are not easily differentiated from other watershed-scale risks, and therefore are largely uncertain. However, given that the estuary- and watershed-scale PCAs suggested that their co-variables largely correlated to the variance explained of their respective models, it appears as though both SBOA and land-uses highly influence estuary-scale eelgrass cover. The literature additionally supports the correlation of land-use to coastal ecosystem health with ecosystem health declines largely attributed to the agriculture industry (Anderson et al., 2006). Moreover, agriculture was found to be the largest anthropogenic land-use of watershed area (mean 46% of non-forested area) as well as being the component that best described the variance explained in the watershed-scale PCA. However, due to high correlations among the watershed-scale dataset, the sensitivity analyses were not performed and thus could not determine the effects onto eelgrass cover. The literature supports that land-based run-off from agricultural area has been demonstrated to negatively affect coastal ecosystems by causing nutrient loading (e.g., nitrogen and phosphorous), which in turn can directly and indirectly cause eutrophication (for regional examples see: Cairns, 2002; Meeuwig et al., 1999; Cairns et al., 2002; Anderson et al., 2006). Eutrophication has been observed in a number of the examined estuaries with Bouctouche and Cocagne having been ranked with “high” levels of eutrophication (Schmidt et al., 2012), as well as reporting the first and third greatest amounts of agriculture from the dataset (NRC, 2012). Nutrient loading and eutrophication are major stressors onto eelgrass, and are the primary cause for regional declines with greater vulnerability and enhanced effects expected in estuaries with low tidal flushing rates (DFO, 2011b). However, filter feeders are able to mitigate the effects of eutrophication by depleting phytoplankton biomass through consumption (Landry 2002, in Cairns et al 2002; Anderson et al., 2006). Therefore, while the watershed land-uses and their stressors pose a significant risk to eelgrass productivity, the presence of SBOA in particularly affected estuaries may in fact be mitigating this risk (Anderson et al., 2006). The ecological impact of SBOA on eelgrass cover at the estuary-scale has thus been described as a “low” risk given eutrophication is a seasonal risk that SBOA can mitigate.

Given that SBOA is potentially mitigating the effects of watershed land-use activities and pressures, there are fewer socio-economic costs associated with the presence of SBOA than compared to its absence. That is, relative to the socio-economic consequences associated with eutrophic estuary conditions (ranked as a “medium” risk), the costs associated to maintain SBOA activity within the estuaries’ ECC limits is a “low” risk. Although additional monitoring and measuring costs should be implicit to ensure maintenance below the ECC threshold, the lack of provincial legislation to mandate such regulations allows for the SBOA industry to incur only routine costs. Moreover, the effects of land-use activity and aquaculture are not effectively managed together, allowing aquaculturalists the risk of exceeding the estuaries’ ECC limits, in turn reducing access to the CRA fisheries through the loss of eelgrass as critical habitat.

To adequately monitor, manage, and mitigate the effects of SBOA onto eelgrass at the estuary-scale, the operational repercussions would classify as a “low” risk given the limited amount of disruption and costs these measures would require. Again, the lack of legislation allows for the SBOA industry to independently allocate its resources, and so a neutral or positive eelgrass response will not cause any disruptions to the SBOA industry.

The strategic repercussions were ranked as a “low-negligible” risk given that there is the propensity for DFO/ Government of Canada to exploit the estuary-scale mitigation effects of SBOA onto land-based pollutions that would otherwise negatively affect critical habitat and fish stocks. Minor to some loss of public trust is expected in light of the sectoral management approaches that fail to include terrestrial pollution sources like agriculture into coastal management planning. However, given that eelgrass may benefit from SBOA at the estuary-scale, and that SBOA is currently regulated at the estuary-scale under the replacement class screening report (2007), there are few if any expected strategic repercussions. The estuary-scale results are summarized below.

Table 7. Risk analysis of the bay-scale effects of SBOA onto eelgrass cover. The statistical analyses have revealed a significant probability between SBOA characteristics and eelgrass cover ($P > 2.16 \times 10^{-16}$), although without better integrating the effects of watershed processes the estuary-scale likelihood of risk was conservatively ranked as a 4 (likely; 76-95% likelihood). The overall impacts and consequences for near-field effects have been ranked as a 2 (low; rounded from 1.86) upon averaging the individual rankings for the ecological, socio-economic, operations, and strategic consequences.

Likelihood: 4. Likely			
Risk Analysis of Impacts and Consequences		Risk Analysis of the Repercussions Tolerability	
Impacts: Ecological <i>2. Low Risk</i>	Consequences: Socio-Economic <i>2. Low Risk</i>	Repercussions: Operational <i>2. Low Risk</i>	Repercussions: Strategic <i>1.5. Low-Negligible Risk</i>
<ul style="list-style-type: none"> ➤ Minor, recoverable short-term (1 year), e.g., seasonal changes in habitat and fish stock 	<ul style="list-style-type: none"> ➤ Reduced access to commercial, recreational, and aboriginal (CRA) fishing activity ➤ Routine cost (mitigation measures) to develop to accommodate habitat requirements 	<ul style="list-style-type: none"> ➤ Limited and temporary disruption of operations, requiring minor to moderate reallocation of resources within the area, regionally or outside of programs ➤ Overspending of the budget \$100,000 ➤ Loss of assets by up to \$100,000 	<ul style="list-style-type: none"> ➤ Some loss of client trust of cooperation, escalating resistance ➤ Some client/public needs go unmet ➤ Sustained negative national media attention / public criticism ➤ Erosion of DFO / Canada's international reputation

The overall bay-scale risks of SBOA onto eelgrass were ranked as a “low” upon assigning each the four categories from 1 to 5 and finding their average. Plotted against the likelihood of the risks, which was ranked as “likely”, the resulting heat analysis map reveals that current conditions are tolerable (Fig. 9) and do not demand immediate ecosystem-based risk management and treatment options.

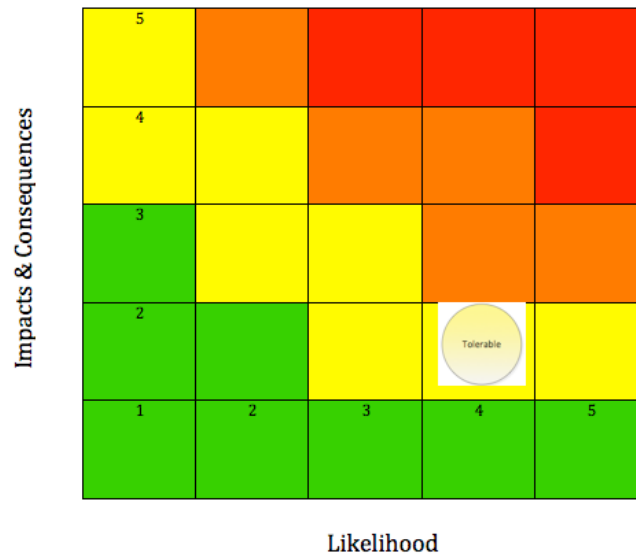


Figure 9. Risk Analysis Heat Map for the estuary-scale effects of SBOA onto eelgrass. Given that the risk analysis results have demonstrated a “likely” likelihood, and that the overall impacts and consequences averaged to be “low”, the estuary-scale eelgrass-SBOA relationship exudes “tolerable” conditions and does not demand immediate ecosystem-based risk management and treatment options.

4.5. Summary of Risk Assessment Results

The risk assessments' results were based on a combination of the available literature, the statistical analyses, and professional opinion and insights. At the local near-field and ZOI scales, the primary management objective should be to protect local eelgrass meadows and conserve their ecological structures and functions as critical fish and invertebrate habitat. In accordance with the literature, the near-field scale was set as causing the majority of direct negative effects onto eelgrass given that a number of disruptive and destructive processes (e.g., shading, smothering, mooring) occur at the near-field scale (e.g., Skinner et al., 2013). As such, the near-field ecological risks were ranked the highest among all of the spatial scales (4, or very high risk) given that the most available research suggests indisputable evidence of long-term ecological damage and non-recovery of eelgrass (Skinner et al., submitted; Skinner et al., 2013), consistent with the "permanently altered or destroyed" criteria described by the amended Fisheries Act (see Koops et al., 2013). The risk analysis concluded not tolerable conditions SBOA conditions at the near-field scale for which management measures are immediately required to intervene.

By comparison, the broader scales (i.e., ZOI, bay, and estuary) have been demonstrated in the literature to provision an apparent mitigation effect through oyster filtration services. These mitigation effects appear to be most pronounced at the ZOI scale such that the manipulated SDI numerical model had suggested that the filtration capacity of SBOA is so significant as to affect phytoplankton concentrations in adjacent bays and the entire estuary (Guyondet et al., 2013). The evidence presented for the ZOI scale would therefore suggest improved ecological conditions for the surrounding eelgrass meadows given that increased filtration services (Guyondet et al., 2013) has improved eelgrass productivity and cover without the direct damaging effects of overlying gears. The ZOI risk analyses had therefore concluded relatively low ecological risk (ranked a 2; low risk); as a result, the risk assessment had revealed extremely tolerable conditions for which additional management measures were deemed not necessary; however, the ZOI-scale should not be excluded from monitoring programs.

At the ecosystem level, the management objectives at the bay and estuary-scales should be to protect entire eelgrass meadows and conserve their valuable ecological

services and functions that make them one of the most valuable ecosystems worldwide (e.g., provision coastal protection, dissolved oxygen, carbon sequestration). At the bay-scale, the statistical results had broadly agreed with the available literature (e.g., Anderson et al., 2006) such that each bay responded dissimilarly to simulated increases and decreases of SBOA co-variables with particular focus on the variable effects of total suspended lease area (TSL) and total aquaculture (i.e., suspended bag and bottom table culture) lease area (TAL) onto eelgrass cover. Although the filtration capacity of cultured oysters can affect the entire bay as well, there is the added ecological risk of SBOA characteristics (i.e., TSL and TAL, as identified in the PCA and SA models) to cause both positive and negative effects onto eelgrass cover, and that these effects are entirely contextualized and bay-specific. Therefore, the ecological risks of SBOA onto eelgrass at the bay scale were ranked higher at 3 (medium risk). Given the contextualized conditions which account for the positive and negative response in predicted eelgrass cover (notwithstanding the unknown long-term effects of filtration onto eutrophication), the bay scale was assessed to have currently tolerable conditions, albeit monitoring measures will be required to determine which bays are adversely affected and to ensure that they remain within their ecological carrying capacity limits.

Broader still is the estuary scale, accounting for watershed processes and therefore land-based nutrient and sediment run-off; cultured oysters are described in the literature as to mitigate these effects, in particular the pressures from agriculture that cause eutrophication (e.g., Meeuwig, 1999). In conjunction with the literature, the estuary-scale statistics had demonstrated that neither the SBOA co-variables nor the watershed land-uses exerted any particularly high influence onto eelgrass cover; either suggesting that the adverse effects of agriculture and other land-based nutrient sources are in fact mitigated by SBOA or that they are not potent enough to elucidate an effect onto eelgrass cover at the estuary-scale. Regardless, the estuary-scale appears to have been either too large of a spatial scale for cumulative pressures to distinctively affect eelgrass cover. However, eelgrass data was representative of only overall eelgrass distribution, and not the quality of structure, functionality, and health of the eelgrass meadows. Nonetheless, the ecological risk of SBOA onto eelgrass was considered to be less at the estuary- than the bay-scale and was therefore ranked as a 2 (low risk). It is

worthwhile to mention that these ecological risks may be either under- or over-estimated depending on the capacity of oysters to exert bottom-up control on phytoplankton, which could cause long-term eutrophication, and therefore ongoing eelgrass and SES degradation (Anderson et al., 2006; McKindsey et al., 2006; Schmidt et al., 2012). The risk analysis had illustrated currently tolerable conditions (again excluding the possibility of adverse long-term effects of enhanced eutrophication), although more research is needed to identify whether cultured oysters are mitigating or enhancing eutrophication along eastern NB and to what degree.

The literature largely focuses on the various ecological effects of SBOA and fails to make the connections to the socio-economic consequences, operational and strategic repercussions (e.g., Cranford et al., 2006; Bastien-Daigle et al., 2007); therefore, these aspects of the risk assessments largely hinged upon the potential ecological risks. It is worthwhile to note that all of these anthropogenic consequences are ranked less than their respective ecological risk, whether or not this assumption is accurate in actuality. It has been argued, however, that adverse damages onto eelgrass would cause a direct loss of access to the CRA fisheries given the loss of critical habitat (e.g., nursery, foraging, and feeding habitat) available to many economically valuable species and their food sources (ICES, 2012; Kenchington et al., 2012). As the collapse of the cod fishery had significant negative long-term effects onto the socio-economic structure of Newfoundland (e.g., Hamilton et al., 2001), the loss of access to the CRA fisheries along eastern NB and the sGSL could have significant repercussions for rural fishery-dependent livelihoods. The ecological and socio-economic collapse exemplified in Newfoundland was sufficient to draw negative international attention and had eroded Canada's international reputation; should such socio-economic consequences accrue along eastern NB and the sGSL in response to the loss of access to the CRA fisheries, in turn due to the loss of critical eelgrass habitat, it can be assumed that such strategic repercussions would also ensue. The strategic risks were found to be highest for the near-field and bay-scales (3, medium risk; 2.5, medium-low risk, respectively) given that this where the largest potential for the cascade of multi-sectoral risks to occur.

The worse the ecological risk, the worse the assumed socio-economic consequences, and in turn the higher risk ranking of negative strategic repercussions;

accordingly, the near-field scale was ranked the highest strategic risk (3; medium risk). The operational consequences, however, remain to be ranked as low or negligible risks across the four spatial scales. Given that the aquaculture industry is responsible for its own management, and because the overarching policies in place only account for sectoral management approaches (e.g., Transport Canada, 2007), there are few direct incentives and consequences onto the SBOA industry to prevent the loss of eelgrass habitat. The SBOA industry is, however, legislated under the Fisheries Act to prevent permanent damages onto valued environmental and socio-economic components (VECs), including habitats that support the CRA fisheries (e.g., eelgrass, Transport Canada, 2007), although given the lack of monitoring programs and mitigation strategies in place, permanent damages can go unnoticed and unsolved, potentially proliferating damages onto other sectors and the SES. Near-field operational risks were ranked the highest (2.5; medium-low) given that any costs that would be accrued onto the aquaculturalist would occur within the immediate confines of the SBOA lease area.

5. Discussion

Comprehensive studies to determine whether SBOA has a net positive or negative effect on coastal ecosystems have thus far been limited (Anderson et al., 2006, McKindsey et al 2006). This project was designed to assess the effects of SBOA onto eelgrass cover at multiple spatial scales along eastern NB, and to conduct a holistic ecosystem-based risk management (EBRM) analysis on each spatial scale's effects. More specifically, the purpose of this research was to: 1) establish the scope and effects of the eelgrass-SBOA relationship at each spatial scale; 2) assess the relationship at each of spatial scale and create a vulnerability profile by using the EBRM framework; and, 3) elaborate upon best practices and suggest ICZM recommendations that will promote the health of eelgrass and the sustainable development of SBOA along eastern NB. Using several statistical analyses in conjunction with the available literature, this project has demonstrated that significant trade-offs exist between the effects of SBOA onto eelgrass at different spatial scales.

The literature supports that that SBOA can have a broad range of effects onto eelgrass and the coastal ecosystem (Newell 2004; Anderson et al., 2006; McKindsey et al

2006; Bastien-Daigle et al 2007) and that these effects can be both positive and negative (Dumbauld et al 2009; Tallis et al., 2009) depending on the scope and scale used. However, as the statistical analyses have clearly demonstrated, the extent and direction (i.e., positive or negative) of these effects are highly contextualized to the sample sites and spatial scales examined (Anderson et al, 2006). The literature review and risk assessments have revealed a complex and non-linear SBOA-eelgrass relationship exists along eastern NB, and that these opposing effects can proliferate at the local (i.e., near-field and zone of influence) to system (i.e., bay, estuary scales) to SES scales (Dumbauld et al 2009).

The most significant negative effects of SBOA were demonstrated at the near-field scale due to the effects of shading, smothering, and physical disturbances (Skinner et al., submitted; Skinner et al., 2013). Using the active SBOA lease (ASL) area as a proxy for similar near-field damages, it is estimated that a minimum of 45.8 ha of eelgrass, or mean 2.3% eelgrass per bay area, is already at risk in the n=14 examined bays along eastern NB. Given that eelgrass is recognized as an ESS and directly linked to the CRA fisheries, such loss is significant for the overall health and integrity of the CRA fisheries, coastal ecosystem, and the SES.

Contrastingly, new SDI numerical modeling by Guyondet and colleagues (2013) has demonstrated the far-reaching effects of the cultured oysters' zone of influence (ZOI), whereby they are able to significantly deplete the Richiboucto estuary and its bays of phytoplankton concentration. Given that light attenuation is the limiting factor for eelgrass growth (Schmidt et al., 2012; Skinner et al., 2013), the immediately surrounding eelgrass beds can benefit from less light-obstructing low phytoplankton concentrations. However, these filtration benefits to eelgrass may only exist in the short-term; currently, there is insufficient literature to support the long-term effects oyster filtration on eutrophication, albeit mitigation or enhancement. Therefore, the SBOA effects on bay- and estuary-scale eelgrass meadows and their relative ecological roles become less certain over time, although SBOA conditions are considered tolerable for the time being.

However, the manipulated SDI numerical model results (Guyondet et al., 2013) found that the oysters' ZOI for Bedec was >3 the total area of SBOA³², while the entire Richiboucto estuary was similarly found to have cultured oysters depleting phytoplankton concentrations $>3x$ faster than primary production could regenerate (Guyondet et al., 2013). Again, it is possible that the excess phytoplankton depletion observed in the Richiboucto could have both positive (e.g., eutrophication mitigation and increased light attenuation) and negative (e.g., eutrophication enhancement by shifting seston community structures) effects on eelgrass, depending on whether short- or long-term consequences are considered. In accordance with the ASC (2013) guidelines, the Richiboucto estuary should be immediately subjected to intense bay-scale management regimes to mitigate this excess phytoplankton depletion, which could have severe and cascading effects onto the surrounding ecosystem.

The results of this study illustrate that the local environment and the SBOA characteristics largely determine the ecological processes and responses to different pressures (including SBOA), ultimately to determine the variability and vulnerability of eelgrass to anthropogenic pressures (Newell, 2004; Anderson et al., 2006; Cranford et al., 2006; McKindsey et al., 2006; Bastien-Daigle et al., 2007). The assessment of tolerability to current SBOA conditions along eastern NB have determined that each of four described spatial scales require varying prescriptions of ICZM measures in order to optimize SBOA effects and mitigate eelgrass declines. The bay- and estuary-scale effects determined by the statistical analyses are summarized below (see section 5.1) as well as the risk assessment results (see section 5.2).

5.1. Bay- and Estuary-Scale Effects

Consistent with the expected weight of effects at different spatial scales (Anderson et al., 2006), the statistical analyses found that the smaller bay-scale described greater significance and higher correlations than at the estuary scale. The greater observed variance among eelgrass cover in response to manipulations suggests that the ecosystem is more sensitive to bay-scale effects than estuary-scale effects. Similar to the results of Wagner and colleagues (2012), it is hypothesized that the

³² TAL was estimated to be 18.7% of bay area while the area of phytoplankton depletion (SDI > 1), or ZOI, was 58% of the bay area; see Appendix 8.4.1.

unique hydrodynamic and oceanographic processes as well as aquaculture characteristics (e.g., cultured oyster stocking densities) present within each bay, that may otherwise be negate at the estuary-scale, may be responsible for the highly contextualized nature of these results.

The statistical results suggest that TAL and TSL have greater varying influences onto predicted eelgrass cover at the bay-scale than at the estuary scale. Moreover, the statistical results suggest that there is a scale-dependent critical threshold between TSL and TAL at the bay-scale; that is, when 6.4 and 8.1% of the bay area are reserved for oyster aquaculture, respectively. These results suggest that there is an inflection point at which the ecological carrying capacity limits are reached and eelgrass cover is predicted to decline in response either an increase in either TAL or TSL, but that the these thresholds are contextualized and highly bay- and estuary-specific. However, is also important to note that the only difference between these two co-variables is the inclusion of active bottom lease area (which was not an initial focus of the current research project given its assumed negligence and declining popularity amongst NB aquaculturalists) in the TAL estimate. Results suggest that bottom culture techniques have a significant influence on eelgrass cover at the ecosystem scale. It is hypothesized that these results are indicative of the local-scale effects of biodeposition and SBOA-source nutrient loading onto the benthos, which is increasing anaerobic sediment conditions and eelgrass degradation (e.g., Vinther et al., 2008), as well as more direct shading and smothering that are known to effectuate significant effects (Skinner et al., submitted; Skinner et al., 2013). Therefore, because the planned 10% of estuary area set for SBOA does not account for bottom-culture techniques and effects (DFO, 2011b), this 10% estuary-area threshold appears too high as it can still allow for negative effects to occur and accumulate.

Unfortunately, much the watershed data was inappropriate for many of the statistical analyses given the lack of data and its format, and it would have been erroneous to make any further conclusions based on the available data. However, it was interesting to find that the PCA determined equal influence from all watershed land-use types, and although it is renowned that agriculture is among the most significant land-based source of water quality degradation (e.g., Cairns, 2002; Cranford et al., 2006), it

was not any more influential than industrial or infrastructure, both of which are almost non-existent in the area. Therefore, it appears as though the available data used for this study is insufficient to determine whether land-based pollutions are a major contributor to the loss of eelgrass along eastern NB, or if its effects are indirectly mitigated by the significant filtration capacity of cultured oysters.

5.2. Risk Assessments

By having plotted the likelihoods of risks against the average ranked risks at each spatial scale, it was possible to determine the tolerability of current SBOA conditions and effects based upon the degree of negative ecological, socio-economic, operational, and strategic consequences. Ultimately, the four risk assessments have revealed significant trade-offs between each of four described spatial scales with varying amounts of prescribed ICZM measures required to mitigate each scale's effects onto eelgrass. It is important to note, however, that the risk assessments had not considered the temporal variability in effects (i.e., short- versus long-term effects) and could therefore be subjected to significant changes in results and recommendations.

The current management conditions and outcomes at the near-field scale have been identified as “not tolerable” given both the high likelihood and consequences of SBOA effects onto eelgrass. The IOM Risk Management Handbook (Cormier et al., 2013) therefore recommends the immediate implementation of management measures in order to treat these risks, ultimately to minimize the effects onto the SES. Contrastingly, the current and contextualized conditions at the ZOI scale were found to be “extremely tolerable” given that the high filtration rates appear to have ultimately allowed for greater eelgrass cover for areas adjacent to SBOA sites in comparison to areas located beyond the ZOI boundaries. Moreover, it was found that the mean effects of SBOA onto eelgrass were “tolerable” at both the bay- and estuary-scales. However, these results are especially variable at the bay-scale, and so although the heat maps suggest that management measures are not immediately required, each bay should be to undergo a required individual assessment to ensure that SBOA has not already achieved or surpassed its ecological carrying capacity and that the coastal SES is not at risk. Moreover, the bay and estuary risk assessment results omit the unknown long-term effects of oyster filtration on phytoplankton communities and whether SBOA enables

eutrophication mitigation or exacerbation; further ecosystem-scale research is needed to distinguish the short- and long-term effects.

5.3. Caveats and Limitations

As with much of the available scientific literature, there are many caveats and limitations to mention. First, the different risk analysis results at each of the spatial scales are not conflicting, but contextualized; the effects of SBOA in the ecosystem are expected to vary depending on the scope of the assessment used. The pathways of effects considered (e.g., direct or cumulative) and the management objectives desired entirely dictate the effects considered for assessment, and in turn the risk analysis results. While the risk assessments for this research project attempted to consider a multitude of possible effects at multiple spatial scales, there were inevitably co-variables omitted from the analyses as well as complex cumulative pressures with unknown net effects. Therefore, the risk analysis results presented are entirely contextualized to the variables considered, and should not be considered out of the context given. Moreover, the results presented can be expected to change with improved scientific understanding of the coastal NB SES, and its complex mechanisms regulating the relationship between SBOA and eelgrass.

There was insufficient literature to support the socio-economic, operational, and strategic repercussions in response to the increase of SBOA along eastern NB. Although SBOA is projected to increase throughout Atlantic Canada in the coming years, it remains to be seen how this will impact the rural communities and livelihoods that depend on the coastal ecosystem and its resources (e.g., CRA fisheries) in the short- and long-term, and in turn affect the local SES. The case of Newfoundland's social structure breakdown in response to the collapse of the cod fishery is an internationally renowned example of a failure to recognize and integrate scientific recommendations into management, resulting in further and more complex SES effects. Should the various levels of government and eastern NB communities wish to prevent such deleterious consequences, further research is required to better understand the interdependence of the eastern NB SES specifically in response to the increase of SBOA to enable improved and more comprehensive risk assessments.

The datasets used to inform the statistical analyses were based upon only the data that was available across all $n=14$ bays and $m=8$ estuaries; that is, important variables that were missing data for some bays were excluded entirely from the analyses (e.g., nitrogen content of eelgrass leaves, relative eutrophication score). Moreover, the proxy for the productivity of eelgrass was limited only to aerial cover, as observed namely by satellite imaging and field surveying; parameters that best represent the health of eelgrass and its ecosystem have been identified (e.g. above-ground biomass, shoot density; Schmidt et al., 2012; McHanon et al., 2013) but omitted from the statistical analyses due to a deficiency in data collection methods. Although the effects of SBOA cannot be considered in isolation but rather as a contributing factor to the cumulative effects onto eelgrass, the relative effects could have been considered if there had been baseline data (e.g., bay and estuary eelgrass cover prior to the installment of SBOA) and/or reference sites (e.g., bays and estuaries without the presence of SBOA) in order to better assess the relative contributions of SBOA habitat effects. A greater number of both eelgrass and SBOA co-variables as well as non-SBOA data to compare to would have increased both the rigidity and statistical power of the analyses. It should be noted, however, that due to the statistically small number of available estuaries to test ($n=8$) meant the analyses were limited to few available degrees of freedom ($DF=7$) and so only a handful of SBOA co-variables could be tested at once ($n=6$) without compromising the statistical validity of the analysis (pers. comm., Joey Hartling, September 14, 2013) hence why there were no interaction terms tested at the estuary scale. It can therefore be argued that the estuary-scale statistical analyses were less statistically powerful and definitive than the bay-scale analyses, meaning that there yet could be an evident interaction between SBOA and eelgrass but there is insufficient data to observe it.

Many of the bays along eastern NB, throughout the sGSL and Atlantic Canada remain to be comprehensively sampled. First, there is simply a significant number of bays and estuaries that have not been scientifically sampled or assessed for eelgrass cover. Second, the bays and estuaries that have been sampled have been done so for various experiments and analyses, and so there is an inconsistency among sampling methods and data. This issue was particularly exemplified when EC and DFO had

individually collected such different eelgrass data (see Table 8 in Appendix 8.1.1.) that it so was incomparable that the chi-square results had demonstrated significant observer biases (see Appendix 8.2.3.2.) and would have represented inconclusive results that would have reduced the validity of the risk assessments. Moreover, the LiDAR results were so significantly variable³³ from the ground-truth data that they could not be included in the statistical analyses given that they would have also reduced the validity of the results. As a result, the 2007-2009 eelgrass field-survey data was used as ground-truth data, but as aforementioned, the field-survey data was disparately collected and accuracy could be vastly improved upon. Therefore, the eelgrass cover date used as the dependent variable for all the statistical analyses was relatively outdated compared to the updated data the LiDAR (2012) had the potential to provide. It is cautioned that eelgrass cover in these embayments has changed during the course of the past four to six years and the effects have since enhanced.

One of the greatest limitations associated with the risk assessment itself. The EBRM framework and the guiding risk assessment structure (Cormier et al., 2013) prevent the inclusion of beneficial and neutral risks to be taken into account. As previously discussed, it is important to demonstrate the positive as well as negative effects of SBOA onto eelgrass, which cannot be expressed in the current risk analysis framework. In addition, the amount of uncertainty associated with any given risk assessment cannot be accurately accounted for; for example, the short- versus long-term effects of oyster filtration on eelgrass are currently inconclusive for eastern NB. Therefore, there is no way of accurately informing the risk assessment of either a potentially positive or uncertain interaction, which can cause high variability in the risk assessment outcomes.

Lastly, many contradictions were found within the literature as to the various positive and negative effects of cultured oysters onto eelgrass and the ecosystem. However, these contradictions are simply indicative of the contextualized nature (e.g., oceanographic and hydrodynamic processes, SBOA stocking densities, nutrient

³³ The LiDAR had been insufficient to examine the eelgrass present within the eastern NB bays and estuaries given that they had been testing in September (2012) when it had been active Hurricane season, resulting in high reflectance of the LiDAR and minimal data collection. It has been recommended to instead use acoustic remote sensing methods for collecting (Cranford et al 2006).

pollution) of each of case studies and sample sites examine; that is, it is expected that each analysis would yield disparate results specific to the examined embayment and the research methods pursued. It is therefore important to consider upon reviewing the literature that all results are specific their respective research project and should not be considered as a standard effect and impact. The ecosystem-based risk assessment can only be appropriately informed with contextualized data, and so it is essential that the aforementioned data collection limitations are better addressed prior to further intensive industry development. Otherwise, it is possible to make inappropriate conclusions and management recommendations should the information be based on generalized research and data³⁴.

5.4. Management Recommendations

As it has been proposed within this research project and throughout the literature, there is an immediate need to further develop and implement an appropriate ICZM plan for eastern NB's coastal and estuary ecosystems (e.g., Turcotte-Lanteigne and Ferguson, 2008a). Although there are several proposed definitions and characteristics of ICZM (see Wiedemeyer, 2010 for examples), it is important to consider the unique SES context eastern NB and the surrounding sGSL while still embodying an EBRM perspective. The implementation of a comprehensive ICZM plan would allow for defined management objectives, greater insight to the achievability and consequences of any given management measures, and reduce the risk of effects from cumulative yet unanticipated pressures by eliminating sectoral management approaches. Fortunately, the SBOA activity along eastern NB is currently regarded as being a low-impact activity within the natural ECC limits of the ecosystems (Cranford et al., 2006); ongoing eelgrass declines suggests that there are cumulative impacts that are affecting the coastal ecosystem. **Monitoring and management of SBOA must therefore consider itself as a contributing factor of the intimately interconnected multiple anthropogenic activities and uses of the coastal zone** (see Appendix 8.4. for pathways-of-effect) that in turn cause complex cumulative and sometimes synergistic effects (Cloern, 2001).

³⁴ For example, Bastien-Daigle and colleagues (2007) suggest that SBOA along the eastern coast of NB is not beyond its regional ECC given the historical natural oyster population; however, given the number of changes to environmental conditions in recent decades, it is inappropriate to assume that the present ECC would be equal to that under natural conditions.

SBOA sustainable development inevitably requires a holistic management perspective to account for cumulative impacts and balance the positive, negative, and neutral effects that accrue at multiple spatial scales within the coastal ecosystem. Because of the inherent complexity of these cumulative effects (e.g., these effects are place- and scale-dependent and are based upon the desired management outcomes; pers. comm., Marc Ouellette, August 20, 2013), the EBRM is best used to offer a structured approach to assess the risks of conserving the valued ecological benefits. Further, ICZM can be used to account for the EBRM assessments and objectives to create effective and efficient solutions to best benefit the coastal SES. It is argued that the sustainable development of SBOA and conservation of eelgrass ecosystems therefore require the both an EBRM approach to determine the relevant risks and an ICZM approach to determine the contextualized resolutions.

The first step towards an ICZM plan is to better **integrate the relevant sectors and stakeholders in order to improve effective communications and governance** of the coastal zone and its dependencies (e.g., resources and resource-users). It has been previously identified by Turcotte-Lanteigne and Ferguson (2008) that various levels and agencies within government must improve communication efforts to better coordinate activities and create cooperative policies and programs that will allow for the sustainable development and management of sGSL coastal zones. Additionally, government must recognize the need to build decision-making capacity at the local community and municipal levels in order to integrate traditional and place-based knowledge that could serve to create contextualized management plans specific to the local SES conditions. Currently, there are several complex multi-jurisdictional and governance overlaps within the sGSL and along eastern NB (DFO, 2007), with much of the management implementation taking place in the absence of capacity at the community level (Turcotte-Lanteigne and Ferguson, 2008a). There are several tenets of community-based co-management that could be better incorporated into the ICZM plans to ultimately promote greater compliance within communities as well as engage the resource-users to help monitor their local environments (*see* Kearney and Berkes, 2007). There is an immediate need to compile data and create a historical ecological baseline, and to integrate spatial social, economic, environmental and traditional knowledge

information. It is the role of stakeholder meetings to identify and support the development of incentives for social and industrial cooperation, collaboration, and governance.

Although citizen science and community-based monitoring programs could be benefited from improved stakeholder communication and integration, it is essential to first **recognize the need for improved monitoring programs and standardized data collection methods**. Monitoring programs that specifically assess cumulative effects within each bay and estuary are necessary; programs tailored specifically to each embayment are thus required (Cranford et al., 2006), in turn demanding the improved cooperation, participation, and coordination of researchers regardless of which stakeholder group they represent. Effective monitoring programs have the propensity to quantify the ecological effects attributed to increased SBOA (GESAMP, 1996) and help develop mitigation measures to minimize the HADD onto eelgrass (Cranford et al., 2006). Moreover, the improved data collection of various benthic and pelagic indicators, as well as including temporal variables (e.g., seasonal and annual processes) can help differentiate between natural variations and effected habitat responses to SBOA. Improved aerial data collection methods with higher resolution (e.g., sonar, acoustics; Cranford et al., 2006) could complete the dataset and provide better information as the current status and ongoing changes to the environmental conditions and eelgrass cover. Also, improved collection methods could reveal more information regarding nutrient, contaminant, and sediment-loading processes; given that these pressures have been identified as the top three risks onto regional eelgrass cover (DFO, 2011a) yet there is currently no government-mandated research to monitor the amounts or effects of these pressures, it can be argued that the status of coastal health could be greatly under- or over-estimated, thus creating difficulties in effectively managing and mitigating the effects. It is therefore of high priority to create detailed and adaptive monitoring programs that integrate various stakeholders to collect and assess data pertinent for understanding the complex eelgrass-SBOA relationship within the context of each embayment's unique SES.

Given that there are several points for improvement within data collection and dissemination, the number of remaining scientific uncertainties demands that **SBOA**

management is approached with precaution so as to not underestimate the cumulative impacts from multiple stressors. The inconclusive estuary-scale results (i.e., similar negligible impact from all tested co-variables) suggests that there is insufficient data to yet determine whether the net SBOA is either be positive or negative. Given the complexity of the ecosystem, a precautionary ICZM approach would reveal that the optimal management intervention point would be through impact prevention and not compensation (pers. comm., Marc Ouellette, August 20, 2013). Preventative ICZM would suggest that while it is possible to have cultured oysters mitigate the effects of eutrophication via filtration services, this should be considered as an additional management benefit rather than a management opportunity. That is, rather than increasing SBOA to mitigate the effects of land-based nutrient and sediment loading, preventative ICZM should consider the contextualized conditions to best control the pressures by regulating the amount of land-based loadings. Even so, it is important to consider that cultured oysters can have adverse effects on phytoplankton communities and could enhance eutrophication; from this perspective, it is possible that some land-based nutrient-loading is now required to support high concentrations of cultured oysters and ultimately prevent SBOA-induced eutrophication. It therefore of outmost importance to comprehensively observe the current environmental conditions and associated cumulative effects both within and around each embayment to best understand the current management needs and then approach management implementation in a precautionary fashion.

Although ICZM should be precautionary to avoid deleterious cumulative ecosystem effects, it is possible to **promote the sustainable development of the SBOA industry through better ICZM planning and SBOA best practices**. While the current status of SBOA along eastern NB and the sGSL has been determined to be an overall low-risk activity (Cranford et al., 2006), several authors have suggested that coordinated oyster aquaculture could be considered a significant ecological benefit (Ulanowicz & Tuttle 1992, as cited by Bastien-Daigle et al., 2007) by increasing the structural and functional diversity and sustainability of the ecosystem (Prins et al., 1997) to ultimately help facilitate the recovery of eelgrass (Newell and Koch, 2004) and promote its productivity. Although SBOA cannot be considered in isolation, as it can be a

contributor to the cumulative impacts that negatively effect eelgrass, there are a number of best practices that can maximize the SBOA benefits while minimizing their damages.

First, it is important to improve near-field conditions so as to minimize the localized negative effects. The SBOA industry should be mandated to utilize less intensive gears (e.g., long-lines, horizontal rope floating rack systems) to reduce benthic shading and smothering effects as well as increase light attenuation (Courtenay et al., in prep.) to promote eelgrass productivity. Additionally, the effects of smothering can be mitigated with strict overwintering and bottom table culture regulations that will minimize the impact onto eelgrass; suggestions include designated areas with an existing absence of eelgrass so as to not create newly impacted zones. Interestingly, the SBOA industry has recently adopted such long-line gears that minimize negative benthic effects through the power of market forces (e.g., quicker to harvest, greater economic value) rather than mandated regulations (pers. comm., Marc Skinner, June 12, 2013). Although the indirect ecosystem benefits associated with the increased socio-economic value of less-intense SBOA practices has occurred in the absence of industry regulations, it does not justify the lack of adaptive management that would otherwise mandate these best practices.

Second, the evident positive effects of SBOA at the ZOI scale should be maximized; as a part of the contextualized monitoring programs and management plans, the optimal SBOA spacing throughout the bay and estuary should be determined in order to maximize the filtration benefits of increased light attenuation. However, because the ZOI of one SBOA site can overlap with another ZOI, the cumulative zone of phytoplankton depletion can account for the entire bay area and its adjacent bays as well. In conjunction with these findings are the results from the statistical analyses, which suggest a greater SBOA influence onto eelgrass at the bay-scale rather than estuary-scale. However, current management practices manage SBOA at the estuary scale by limiting the area of SBOA lease to no more than 10% of the estuary area (DFO, 2011b). Due to the recent research and results, **it is recommended that the scope of management efforts are down-scaled from the estuary- to the bay-scale** given the number of unique processes that dictate the eelgrass-SBOA relationship that differ from bay to bay within an estuary. This recommendation is additionally supported by the

Richiboucto seston depletion results (Guyondet et al., 2013) that suggest that oysters are filtering significant amounts of phytoplankton and therefore have the propensity to alter coastal food-webs and affect long-term eutrophication. Despite the relatively low densities of cultured oysters along eastern NB than those reported along the eastern coast of USA (e.g., Newell and Kock, 2004; Wagner et al., 2012), it is important sustainably develop SBOA to avoid a significant spatial competition effect at the bay-scale. Although it is still important to consider the estuary and its watershed for cumulative ecosystem-scale effects, it is feasible that eelgrass productivity and ecosystem health could be improved by sustainably managing SBOA at the bay-scale. Through the combined efforts of shifting SBOA practices to less intensive gears, better spacing, and bay-scale ICZM plans, it is proposed that the SBOA industry along eastern NB can be sustainably developed and offer coastal SES benefits.

6. Conclusion

Like many coastal and estuarine zones, the eastern coast of NB along the sGSL has been identified as having a unique and complex social-ecological system (SES) that can only be effectively managed with ICZM (DFO, 2005). However, the ongoing decline in regional eelgrass, a recognized ecologically significant species (ESS) and indicator of ecosystem health (DFO, 2009), suggests that there are still untreated risks that threaten the health and functionality of the coastal ecosystems and its interconnected SES. As a multiplicity of anthropogenic activities have created cumulative effects onto the coastal ecosystem, it has become increasingly difficult to discern the contribution of these effects in isolation. As suspended bag oyster aquaculture (SBOA) continues to become an increasingly popular coastal zone use, its relationship with eelgrass will continue to evolve as new pressures accrue. Using both a comprehensive literature review of the sGSL and raw data supplied by a number of governmental agencies, this research project aimed to discern the relationships and risks of SBOA onto eelgrass at multiple spatial scales along eastern NB and to offer ICZM recommendations and resolutions to promote the sustainable development the SBOA industry and conservation of eelgrass benefits and SES integrity.

This research project has approached the SBOA assessments with an EBRM framework to determine how SBOA interacts with eelgrass within the local coastal environment, and what are the effects that can occur at different spatial scales. It was found that cultured oysters have significantly different ecological effects at each distinct spatial scales, with positive, negative, and neutral effects occurring at each of the near-field, zone of influence (ZOI), bay, and estuary scales. Next, the research project analyzed which SBOA co-variables were most influential to eelgrass cover at the various spatial scales, and what were the observable critical thresholds at the bay- and estuary-scales. The literature had suggested a variety of responsible SBOA characteristics, including the SBOA stocking densities (Cranford et al., 2006; Wagner et al., 2012; Skinner et al., 2013), and SBOA lease age (Skinner et al., 2013). The statistical analyses found that total SBOA lease (TSL) area and total oyster aquaculture lease (TAL) area (e.g., $TAL = TSL + \text{bottom table oyster aquaculture}$) were most responsible for the eelgrass variance explained at the bay-scale, but that the estuary-scale co-variables (both SBOA and watershed) yielded indifferent results and suggested negligible impacts. The statistical results suggest that there is a scale-dependent critical threshold between the bay-scale TSL and TAL, or 6.4 and 8.1% of the bay area being reserved for oyster aquaculture, respectively. Therefore, because the planned 10% of estuary area set for SBOA does not account for bottom-culture techniques and effects (DFO, 2011b), this 10% estuary-area threshold appears too high as it can still allow for negative effects to occur and accumulate. More research is needed to validate these results and offer explanations as to the mechanisms driving this critical threshold.

Using the available literature as well as the statistical results to inform the risk assessment, it was generally found that the risk of SBOA to cause negative effects onto eelgrass was relatively low across the ZOI, bay- and estuary-scales (tolerable conditions) but relatively high at the near-field scale (not tolerable conditions). As a result, several management resolutions were offered specifically for the near-field scale, although many general ICZM recommendations were presented to promote the sustainable development of the SBOA industry and protect eelgrass ecosystems at large. These recommendations included: 1) consider SBOA as a contributor to the cumulative coastal impacts, not in isolation; 2) integrate sectors and engage stakeholders for

effective communication and governance; 3) improve monitoring programs and standardize data collection methods; 4) use precautionary and adaptive management approaches; 5) facilitate better ICZM planning and mandate best practices, like less intensive gears, better spacing, and bay-scale ICZM plans. Given the propensity to improve the ecological, social, and economic conditions along the sGSL upon implementing an ICZM plan, it is hoped that these management recommendations will influence governmental regulations and policies by legislatively mandating that ICZM becomes immediately implemented throughout the sGSL.

Although the literature ascertains that SBOA along the eastern NB coast represents a relatively low-impact activity and persists within the natural ecological carrying capacity (ECC) of the sGSL's bays and estuaries (Cranford et al., 2006; Bastien-Daigle et al., 2007), this research project has determined that residual and cumulative risks still threaten critical eelgrass habitat and ecosystems. Therefore, it was concluded that contextualized ICZM measures are absolutely required in order to achieve the long-term sustainable development of SBOA and improve coastal SES health along eastern NB and the sGSL. These recommended ICZM measures are available within DFO's current regulatory mandate, and despite the recent amendments to the Fisheries Act and disposal of the RCSR, are still achievable. In fact, it will become increasingly important to consider the aforementioned recommendations as the government moves forward in planning and implementing an ecosystem-based ICZM approach in order to achieve the regional and national goal of sustainable development within the coastal zone.

7. References

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8. Appendix A

8.1. Methods

The current project was initially intended to be a continuation of the research currently underway at the DFO Gulf Fisheries Center (Moncton, NB) under the Program for Aquaculture Regulatory Research (PARR) Shellfish Aquaculture Ecosystem Carrying Capacity Research (8Ai). The PARR project proposal was accepted in August 2011 with the purpose to identify and examine the relationship between bivalve aquaculture density (a proxy for bivalve filtration), eelgrass coverage (a proxy for productivity), and depth distribution (a proxy for water transparency) at a bay-wide scale. However, complications with the LiDAR methods caused for significant data collection issues and inconsistent results relative to the available ground-truth data. With the exception of the LiDAR images, the 8Ai PARR results were not included for analyses within the current project.

8.1.1. Bay and estuary scale data

It is worthwhile to note that during the EC and DFO sampling and data collection, different methodologies were employed from estuary-to-estuary and year-to-year, resulting in separate and non-compatible eelgrass classification categories. Moreover, only one of the contributory bays in both Tracadie and Shippagan estuaries were sampled for eelgrass cover; data for Tracadie South, Shippagan North and Shippagan Inlet were not provided. Table 8 summarizes the different methodologies used by EC and DFO for eelgrass collection.

Table 8. A summary of the Environment Canada and Fisheries and Oceans Canada eelgrass data collection methods for eastern NB estuaries (n=8) from 2007-2009. Classification categories: GQ = Good Quality, MQ = Medium Quality, PQ = Eelgrass absent/Poor Quality; DE = Dense Eelgrass, ME = Moderate Eelgrass, TE = Thin Eelgrass, EE = Exposed Eelgrass; EP = Eelgrass Presence. Table adapted from AGRG, 2012; St-Simon was added. Adapted from the AGRG (2012) report with additions.

Estuary	Year Sampled	Aerial photography (EC)	Quickbird satellite (EC)	# Field survey sites (DFO)	Side-scan sonar/video camera (EC)	Classification categories (EC/DFO)
Miscou	2009	X		103		GQ, MQ, PQ
St-Simon	2007		X	123		DE, ME, TE, EE
Tracadie *	2009	X		94		GQ, MQ, PQ
Richibucto	2007		X	180	X	GQ
Bouctouche	2009	X		688		GQ, MQ, PQ
Shippagan**	2007		X		X	DE, ME, TE, EE
Néguac	2009	X		123		GQ, MQ
Cocagne	2008	X		405	X	EP
Tabusintac	2008	X		283	X	EP

* Note: Tracadie is represented here only by Tracadie North and the surveys omit Tracadie South eelgrass cover data, which had to be accounted for with a predictive statistical analysis.

** Note: Shippagan was omitted from statistical analyses due to the lack of DFO field surveys and therefore related SBOA data.

8.1.2. Watershed Data

Table 9. List of Natural Resources Canada (NRC) abbreviations and descriptions of primary land-uses of non-forested areas commonly found within each watershed in eastern New Brunswick.

Value	Description
AGR	Land primarily used for growing agricultural products or non-timber tree products as well as fields and pasture land
DND	Land primarily used for National Defense training and exercises
IND	Land primarily used for industrial purposes including their processing facilities
INF	Land primarily used for transportation, communication and/or utilities
REC	Land primarily used for sport, recreational, cultural and/or entertainment activities
SET	Land primarily used for urban or rural residential purposes
WIL	Land that is incapable of growing trees and uninfluenced by human activity
	Blank

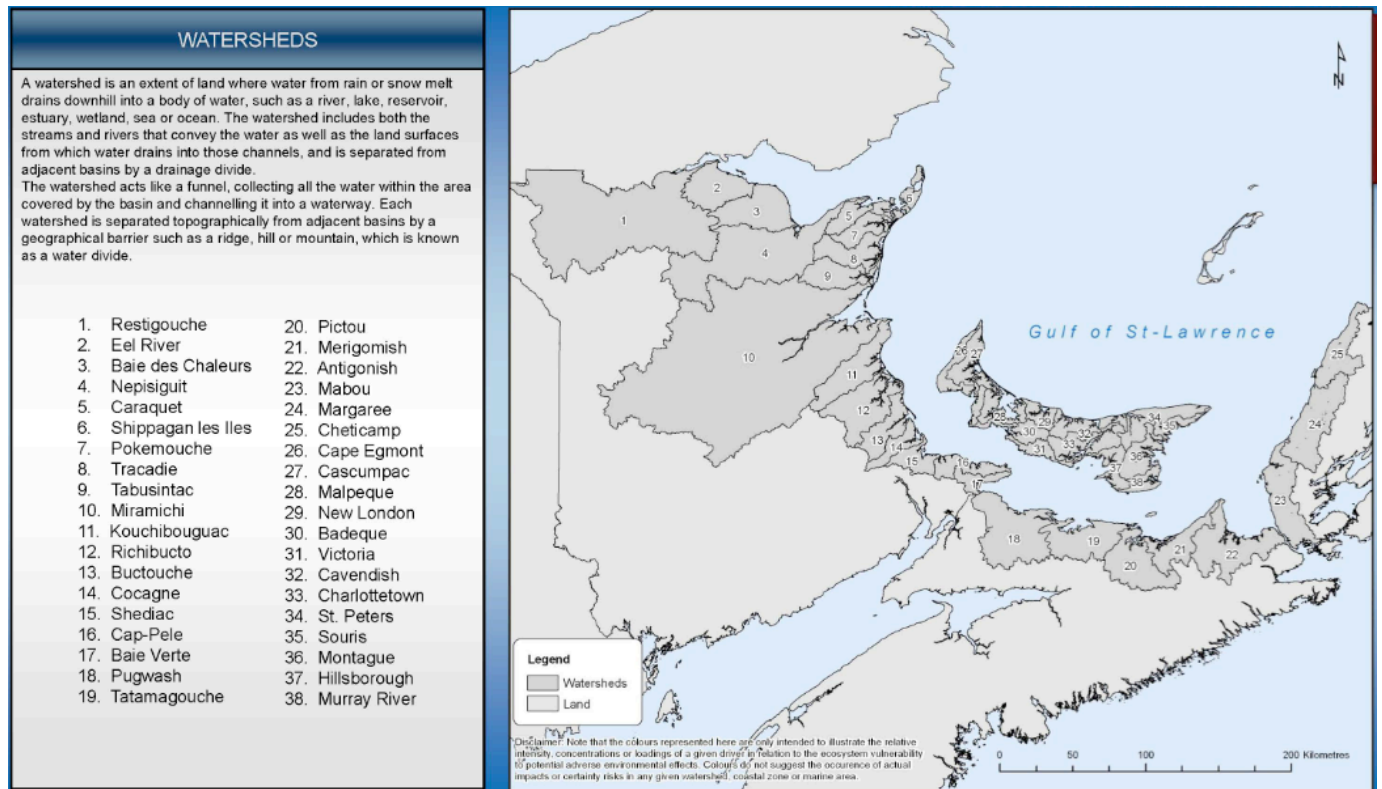


Figure 10. Map of the $n=38$ watersheds that influence the southern Gulf of St Lawrence. Watersheds 1-17 are within the provincial boundaries of New Brunswick; therein, watersheds 5-6, 8-9, and 12-14 influence the $n=8$ estuaries examined for the purpose of the current research project.

8.1.3. Data and Statistical Analyses

The bay, estuary, and watershed variables were provided in either raw area units (e.g., active SBOA lease area, [ha]) or absolute counts (e.g., bag count). Prior to statistical analyses, the datasets were cleaned and significant correlations (>0.60) were removed. Recreational ($n=2$) and wilderness ($n=4$) area were dropped from the dataset due to the lack of sufficient available data for analyses³⁵. A correlations function on the bay-scale dataset revealed that the SBOA biomass was highly correlated to SBOA bag counts, and so in order to prevent models from yielding skewed results due to high correlations (pers. comm. Stu Carson, June 17, 2013) only the SBOA bag counts were included in subsequent analyses. The correlations function additionally revealed that the majority of the watershed land-uses were largely correlated to one another given that the land-use areas are not mutually exclusive (e.g., more forest area results in less non-forested area, more agriculture area results in less settlement area). As a result, the

³⁵ Independent variables require a minimum of five values in order to achieve high statistical power and confidence in the models' results (pers. comm. Joey Hartling, June 16, 2013).

watershed variables were limited to only PCA analyses given that they would have otherwise violated regression assumptions of no correlation amongst independent variables and in turn would have yielded skewed results (pers. comm. Joey Hartling, October 9, 2013).

Table 10. List of raw dependent and independent data used for the bay-scale statistical analyses encompassing the n=14 bays along eastern NB. ASL = Active suspended lease area; TSL = total suspended lease area; TAL = total aquaculture lease area.

Bay	Eelgrass Cover (ha)	Bay Area (ha)	ASL (ha)	TSL (ha)	TAL (ha)	Bag Counts	Gear area (ha)
Miscou	2234	3237	8.33	227.01	318.523	8970	0
St-Simon Inlet	648	1036	16.2	61	74.75	14000	0.039
St-Simon North	613	830	30.08	83.8	109.8	13778	0.156
St-Simon South	668	956	71.16	228.16	258.96	59250	1.291
Shippagan South	761	2835	13.89	279.48	279.479	1300	0.68
Tracadie North	1087	2162	89.56	91.58	91.5827	17981	0.41
Tracadie South	1944*	971	2.42	9.86	14.19	1650	0.005
Tabusintac	1326	3104	58.02	115.39	168.61	11764	0.514
Neguac	1875	3813	75.17	132.25	260.607	32899	0.508
Aldouane	440	690	50.99	64.19	75.89	26500	0.96
Richiboucto Harbour	1030	1726	56.82	100.02	151.32	14200	0.526
Bedec	467	649	56.63	67.12	91.02	31464	1.029
Bouctouche	1719	3023	44.56	132.25	150.851	25098	0.77
Cocagne	935	2014	67.03	138.27	153.5714	13008	0.586

*Note: The predicted eelgrass cover for Tracadie South was estimated to be greater than its respective bay area.

Table 11. List of raw dependent and independent data used for estuary-scale statistical analyses encompassing the n=8 estuaries along eastern NB. Table A) n=6 estuary data and B) watershed data. ASL = Active suspended lease area; TSL = total suspended lease area; TAL = total aquaculture lease area. AGR = agriculture area; IND = industrial area; INF = infrastructure area; SET = settlement area.

A	Eelgrass Cover (ha)	Estuary Area (ha)	ASL (ha)	TSL (ha)	TAL (ha)	Bag Counts	Gear area (ha)
Miscou	2234	3237	8.33	227.01	318.523	8970	0
St-Simon	1929	2822	117.44	372.96	443.51	87028	1.486
Tracadie	3031	3133	91.98	101.44	105.7727	19631	0.412
Tabusintac	1326	3104	58.02	115.39	168.61	11764	0.514
Neguac	1875	3813	75.17	132.25	260.607	32899	0.508
Richiboucto	1937	3065	164.44	231.33	318.23	72164	2.515
Bouctouche	1719	3023	44.56	132.25	150.851	25098	0.77
Cocagne	935	2014	67.03	138.27	153.5714	13008	0.586

B						
Estuary	AGR (ha)	IND (ha)	INF (ha)	SET (ha)	Forested Area (ha)	Watershed Area (ha)
Miscou	723	563	119	768	12122.5	14295.5
St-Simon	719	561	371	1386	4407.5	7444.5
Tracadie	4551	225	198	1493	65451.6	71918.6
Tabusintac	1339	416	161	585	20608	23110
Neguac	789	233	151	828	30789.9	32790.9
Richiboucto	3945	622	1266	940	60102	67272
Bouctouche	8079	278	1106	2242	51297	63014
Cocagne	2926	106	389	1421	28608	33464

8.1.3.1. Predicted Eelgrass Cover Analysis

First, a data frame encompassing the n=6 bay-scale raw co-variables (i.e., ASL, TSL, TAL, bag counts, gear area, and bay area) was created. Next, the data frame was applied to a non-linear beta model³⁶, the generalized linear model (GLM), with a Poisson distribution link given that it most accurately accounts for count and area data. Several models were tested; models with identity (AIC = 284.43, R²= 96.6) and square root (AIC= 223.57, R²=98) links yielded greater AIC scores and lower R² values than the log link (R²; AIC; *P* < 0.05) meaning that the log link offered the greatest amount of statistical power to the model.

The summary report of the GLM described the full data frame as highly significant (*P* = 9.58e-07 to < 2e-16) and that the model itself highly correlated to the variables (AIC=184.48; R²= 0.99). As a precaution to ensure that the model could not gain more statistical power by dropping the least significant independent variable(s), a backward multiple regression analysis was performed. However, the results were identical to the GLM model, indicating that the model required all n=11 variables to explain variation in eelgrass cover. Then, the eelgrass area for all bays was reset to 0 ha, allowing the GML model to predict eelgrass cover; the results are presented in comparison to DFO/EC ground-truth data and the percent difference was calculated³⁷. Validity is added to the model due to the low average percent difference (4.44 %) between the predicted and ground-truth eelgrass cover.

³⁶ It is assumed that the response of eelgrass to SBOA would be best described with non-linear interactions.

³⁷ Percent difference was calculated with the equation: (Known eelgrass – Predicted eelgrass)/Known Eelgrass * 100

8.1.3.2. Legitimacy of Eelgrass Data

A chi-square test was used to assess the variance of eelgrass cover, density, and quality across the n=8 EC/DFO sampled estuaries. It was hypothesized that estuaries with similar SBOA conditions would find that the eelgrass cover, density, and quality would deviate from the mean similarly. However, results had indicated that each estuary was significantly different from one another with no two estuaries having similar eelgrass characteristics. These results suggest that the differences in eelgrass density and quality are indicative of inconsistent sampling methods and classification schemes rather than a true difference in estuaries and their conditions. Moreover, eelgrass quality and density were found to be both positively and significantly correlated to the number of sample sites per estuary, suggesting that the data collection may have been inherently biased (e.g., biased due to various crews, methods, and classification schemes). Eelgrass cover was found to have less correlation to sample sites, and because the data was largely collected by aerial photography as well as field surveys, greater validity was given to eelgrass cover than density and quality. Therefore, eelgrass density and quality were omitted from subsequent analyses while cover was used as the independent variable for all subsequent statistical analyses from which the results were used to inform the risk assessment.

8.1.3.3. Sensitivity of Eelgrass Cover Analysis

The sensitivity of eelgrass cover to each co-variable was tested by having manipulated each variable individually at both the bay- and estuary-scales and then predicted the eelgrass response for each test. A sensitivity analysis (SA) was devised for the purpose of this project and entailed predicting eelgrass cover in response to every manipulated raw data co-variable, excluding bay and estuary area, for a total of n=5 bay and n=4 estuary³⁸ co-variables tested. These co-variables were manipulated to create a new theoretical minimum and maximum value for each variable tested, thus resulting in n=10 bay and n=8 estuary-scale manipulations with a test for each. First, eelgrass cover was re-set to be 0 ha for every bay and estuary in order to allow for the MRA model to predict the eelgrass response. The co-variables' means were calculated and divided by

³⁸ Gear area was omitted from the estuary-scale sensitivity analyses given that the estuary-scale MRA model had dropped it due to the lack of significance.

their respective bay and estuary areas to create a normalized mean per embayment area (expressed as a percent) (see Table 12 below for raw and normalized means)³⁹. These normalized means were then multiplied and divided by their respective raw co-variables to create theoretical maximum and minimum values, respectively, to represent a realistic change (i.e., growth and reduction) in industry production rates⁴⁰. In the case of bag count, which was expressed as an absolute number rather than an area unit, its normalized mean was calculated as the difference between the 2007-2009 and 2011 bag count, then divided by the 2011 bag count (expressed as a percent). Similar to the other normalized means, the normalized bag count was then multiplied and divided by the raw field survey observations to create theoretical minimum and maximum bag count for embayment.

Table 12. The calculated raw and normalized means (%) of each tested SBOA co-variable used for the sensitivity analyses. Bay-scale analyses examined n=5 co-variables for a total of n=10 sensitivity tests; estuary-scale analyses examined n=4 co-variables as it had omitted gear area means, thus for a total of n=8 sensitivity tests.

SBOA Variable	Bay-Scale Means	Bay-Scale Normalized Means (%)	Estuary-Scale Means	Estuary-Scale Normalized Means (%)
Basin Area	1931.9	/	3026.4	/
ASL Area	45.8	2.37	78.4	2.59
TSL Area	123.6	6.40	181.4	5.99
TAL Area	157.1	8.13	240.0	7.93
Bag Count	19418.7	8.46	34480.9	1.92
Gear Area	0.5	0.03	/	/
Eelgrass Area	1124.8	58.22	1873.25	61.89

Each of the manipulated co-variables (i.e., n=10 bay and n= 8 estuary minimum and maximum values) were individually loaded into its' own data frame then tested with the MRA model. The MRA model had predicted the new eelgrass cover in response to

³⁹ Neither the bay nor estuary area was divided by its own area, and were therefore omitted from being manipulated in the sensitivity analysis. Therefore, the analysis proceeded with n=5 bay and n=4 estuary-scale SBOA variables that were manipulated.

⁴⁰ These estimates are theoretical because they do not reflect each embayments' unique processes that would otherwise inform the minimum and maximum ecological carrying capacities for each of the n=5 bay and n=4 estuary SBOA co-variables. Therefore, these minimum and maximum values are arbitrary estimates that are based upon their respective means to offer a consistent method for estimating an increase and decrease in SBOA production.

each of the manipulated co-variables, then graphed as histograms to better observe the patterns resulting from the analyses. In order to ensure that each analysis kept all other variables constant (i.e., true to the ground-truth data) and test only one manipulated co-variable per model, the eelgrass cover was set to 0 ha before each test, thus making it possible to examine the specific bay- and estuary-scale eelgrass responses to the manipulated SBOA co-variables. Upon having plotted the SA results, the visual interpretation of the results was guided by including the lines of moving averages (LMAs) across bays. The LMAs offered a rapid assessment of how the manipulated SBOA co-variables had increased or decreased predicted eelgrass cover relative to the ground-truth EC/DFO data. Moreover, the patterns of the LMAs (i.e., similar or dissimilar to the EC/DFO LMA) were used to infer whether manipulations had caused a significant departure from the ground-truth data, and if so, to what degree and in what direction (i.e., positive or negative influence on eelgrass cover).

If the minimum co-variable tested should yield an increase in predicted eelgrass cover, or if the maximum co-variable yields a decrease in predicted eelgrass cover, then the results would suggest that the current SBOA conditions are at or beyond the ecological carrying capacity (ECC) of that embayment. Inversely, if the maximum co-variable tested yielded an increase in predicted eelgrass cover, or if the minimum co-variable yields a decrease in predicted eelgrass cover, then the results would suggest that the current SBOA conditions are within the ECC limits of that embayment.

Table 13. Summary of the hypothesized ecological implications of the eelgrass cover responses to a range of manipulated SBOA variables.

		<i>Eelgrass Response</i>	
		Eelgrass Cover Increase	Eelgrass Cover Decrease
<i>Manipulated Variables</i>	Minimum SBOA Value	At or Beyond ECC Limits	Within ECC Limits
	Maximum SBOA Value	Within ECC Limits	At or Beyond ECC Limits

8.1.4. Risk Assessment Criteria and Examples

Table 14. Risk management likelihood criteria for the n=5 likelihood categories. Adapted from Cormier et al., 2013.

5. Almost Certain	More than 95%	Occurs regularly here.
4. Likely	76 – 95%	Has occurred here more than once, or is occurring to others in similar circumstances.
3. Moderate	25 – 75%	Has occurred here before, or has been observed in similar circumstances.
2. Unlikely	5 – 24%	Has occurred infrequently before to others in similar circumstances, but not here.
1. Rare	Less than 5%	Almost never observed. May occur only in exceptional circumstances.

Table 15. Risk management criteria for each of n=5 risk categories that describe the degree of the ecological impacts, socio-economic consequences, and operational and strategic repercussions. Adapted from Cormier et al., 2013.

Extreme: A disaster with the potential to lead to permanent or long-term damage to the ability of the department to achieve its objectives.			
Impact: Ecological	Consequence: Socio-Economic	Repercussions: Operational	Repercussions: Strategic
<ul style="list-style-type: none"> -Extinction of an entire species -Permanent, large scale loss of fish habitat 	<ul style="list-style-type: none"> -Long term prohibition to a fishery, including potential negative impacts on other fisheries -Closure of a fishery with direct socio-economic damage to the community -Permanent injunction on development activity 	<ul style="list-style-type: none"> -Inability to operate for multiple weeks, e.g., due to massive loss in resources/staff reductions or massive protests -Major loss/over-spending of budget by more than \$ 5 million, or more than 30% of Branch budget -Major loss of assets \$ > \$5 million -Deaths of 5 or more people -Loss or disclosure of highly sensitive data 	<ul style="list-style-type: none"> -Sustained, long term loss of client / public trust -Canada's international commitments abrogated -Most departmental objectives not met
Very High: A critical event that with proper management can be endured by the organization.			
Impacts: Ecological	Consequences: Socio-Economic	Repercussions: Operational	Repercussions: Strategic
<ul style="list-style-type: none"> -Major shift, e.g., introduction of invasive species, in species composition -Minimum limit reference point is reached -A species is listed as endangered or extinct, recovery plan required -Reduction in genetic diversity of a species -Significant damage to critical fish habitat with limited ability to recover 	<ul style="list-style-type: none"> -Impending closure of a commercial fishery and recreational access in jeopardy, or aboriginal fisheries reduced -CEAA panel review required 	<ul style="list-style-type: none"> -Inability to operate for one week -Disruption of operations, requiring long-term or permanent reallocation of internal resources nationally or outside of program area -Over-spending of budget by \$5 million or more than x% of budget -Loss of assets valued at \$5 million -Deaths of up to 5 people -Loss of key corporate memory 	<ul style="list-style-type: none"> -Significant loss of client trust or cooperation, sustained resistance, e.g., inability to mount joint ventures with clients -Minister reverses major decision due to public or industry pressure -Federal inquiry -Considerable erosion of DFO's / Canada's international reputation -Failure to achieve a key departmental or government objective
Medium: A significant event that can be managed under normal circumstances by the organization.			
Impacts: Ecological	Consequences: Socio-Economic	Repercussions: Operational	Repercussions: Strategic
<ul style="list-style-type: none"> -Species is under significant pressure or at historic lows -Moderate impact to fish habitat, with longer term (3-5 years) for recovery 	<ul style="list-style-type: none"> -Significant reduction to access, e.g., introduction of capacity reduction measures/significant change to how fishing is allowed, e.g., closure of geographic area. -Moderate cost (mitigation/monitoring/full CEAA) to developer to accommodate habitat requirements 	<ul style="list-style-type: none"> -Inability to operate for one or two days -Disruption of operations, requiring reallocation of internal resources regionally or outside of programs -Over-spending of budget by \$1 million or more than x% of budget -Loss of assets valued at- \$1 million 	<ul style="list-style-type: none"> -Some loss of client trust or cooperation, escalating resistance -Some client / public needs go unmet -Sustained negative national media attention / public criticism -Erosion of DFO's / Canada's international reputation
Low: An event, the consequences of which can be absorbed but management effort is required to minimize the impact.			
Impacts: Ecological	Consequences: Socio-Economic	Repercussions: Operational	Repercussions: Strategic
<ul style="list-style-type: none"> -Minor, recoverable short term (1 year), e.g., seasonal, changes in fish stock or habitat 	<ul style="list-style-type: none"> -Reduced access to recreational and commercial fishing activity -Routine cost (mitigation measures) to developer to accommodate habitat requirements 	<ul style="list-style-type: none"> -Limited and temporary disruption of operations, requiring minor reallocation of resources within the area -Over-spending of budget by up to (\$100,000) or more than x% of budget Loss of assets by up to \$100,000 	<ul style="list-style-type: none"> -Some loss of client trust or cooperation, escalating resistance

Negligible: An event, the consequences of which can be absorbed through normal activity.			
Impacts: Ecological	Consequences: Socio-Economic	Repercussions: Operational	Repercussions: Strategic
-Minimal change, falling within normal fluctuations or circumstances, with no impact on stock or habitat -Habitat change completed within acceptable guidelines	-Minor reductions in commercial fishing activity -Minor cost to developer to accommodate habitat requirements	-Limited or no disruption of program operations, with no reallocation of resources	-Minor but recoverable set-back in client / public trust

Table 16. An example of the risk analysis for coastal eutrophication case study based upon the risk likelihood and assessment criteria (see Table 14 and 15, above). Adapted by Cormier et al., 2013; see for more examples of risk assessments.

Likelihood: 5. Almost Certain			
Risk Analysis of Impacts and Consequences		Risk Analysis of the Repercussions Tolerability	
Impacts: Ecological	Consequences: Socio-Economic	Repercussions: Operational	Repercussions: Strategic
-Major shift, e.g., introduction of invasive species, in species composition -Minimum limit reference point is reached -A species is listed as endangered or extinct, recovery plan required -Reduction in genetic diversity of a species -Significant damage to critical fish habitat with limited ability to recover	-Significant reduction to access, e.g., introduction of capacity reduction measures/significant change to how fishing is allowed, e.g., closure of geographic area. -Moderate cost (mitigation/monitoring/full CEAA) to developer to accommodate habitat requirements	-Limited and temporary disruption of operations, requiring minor reallocation of resources within the area -Over-spending of budget by up to (\$100,000) or more than x% of budget Loss of assets by up to \$100,000	-Some loss of client trust or cooperation, escalating resistance -Some client / public needs go unmet -Sustained negative national media attention / public criticism -Erosion of DFO's / Canada's international reputation

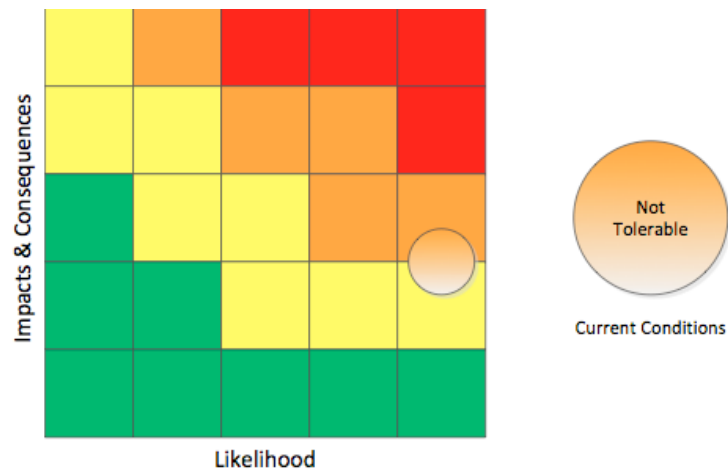


Figure 11. An example of the risk heat map indicating non-tolerable conditions in response to the coastal eutrophication risk assessment case study (see Fig. 11, above). Red blocks are indicative of extremely not tolerable conditions; orange is not tolerable; yellow is tolerable; and, green is extremely tolerable. See the IOM Risk Management Handbook (Cormier et al., 2013) for more examples of heat maps.

8.3. Pathways-of-Effects

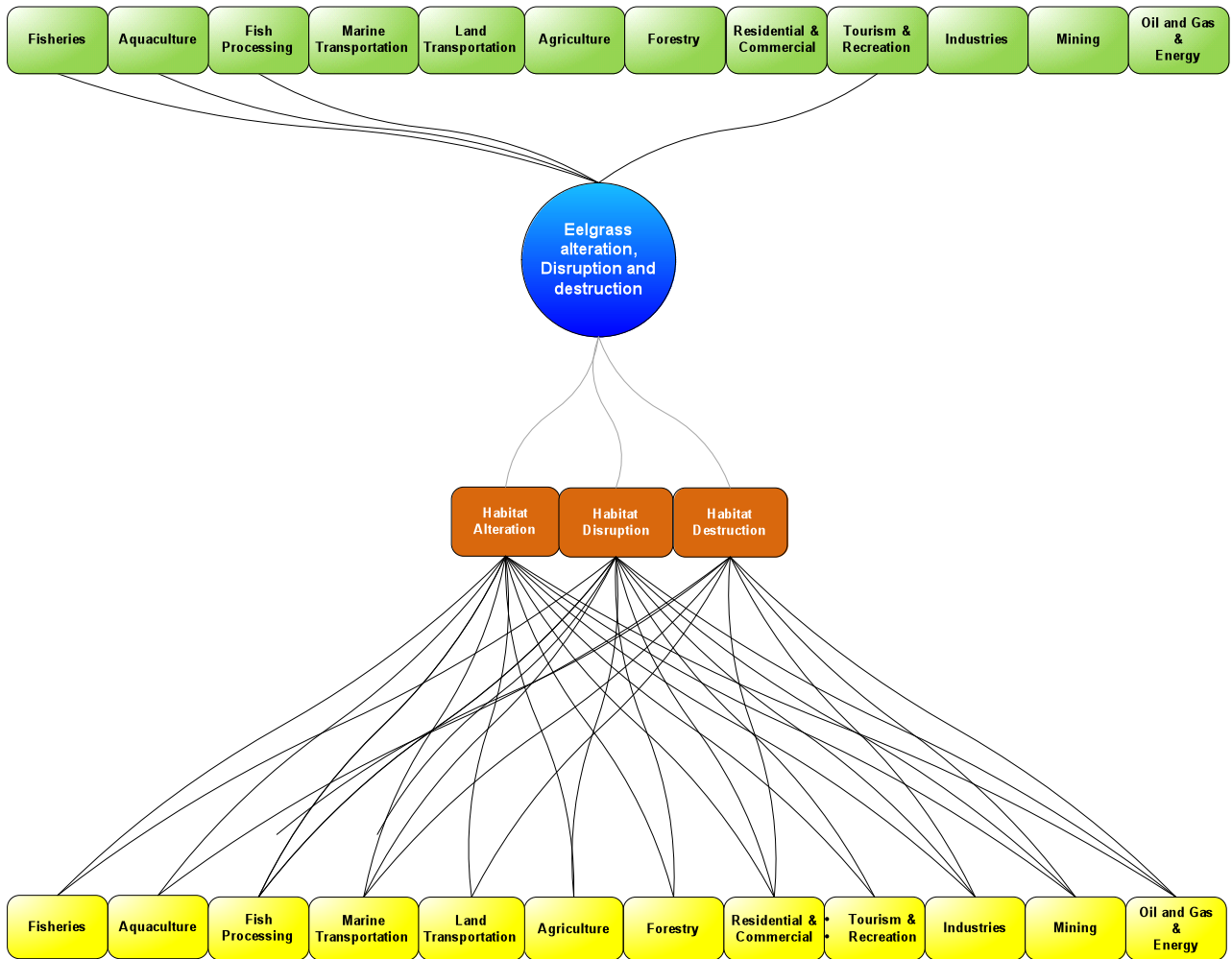


Figure 12. A conceptual model of the various cause-effect pathways for HADD onto eelgrass. The yellow nodes represent the cumulative disturbances and pressures caused by the multiple competing human activities and uses; the orange nodes are the stressors and impacts; the blue node is the response of HADD onto eelgrass; the green nodes are the socio-economic sectors that experience repercussions due to the loss of eelgrass.

8.4. Results

8.4.1. Manipulated SDI Numerical Model

The numerical model created by Guyondet and colleagues (2013) to examine the dynamics of phytoplankton depletion caused by SBOA in the Richiboucto estuary was altered to provide new results relevant to this study. The seston depletion index (SDI) was used to examine the area and degree of phytoplankton depletion and accumulation as it responded to different SBOA conditions. First, the model predicted that the absence of SBOA anywhere in the estuary would cause 100% accumulation of phytoplankton

throughout the entire estuary, even in well-mixed areas like the estuary's mouth in Richiboucto Harbour (see Fig. 15 for water renewal times). This result suggests that SBOA significantly influences and regulates the amount of phytoplankton present within the estuary, and that seston depletion is predicted only in conjunction with SBOA presence.

Next, the model was manipulated to restrict the presence of SBOA sites to within only Aldouane and only Bedec, and results were compared to results from SBOA present through the entire estuary in order to examine the scale and effect of the ZOI. Despite the relatively slow water renewal time for both of these bays, results indicate that the ZOI is not limited to the bay in which SBOA was modeled to be present (Fig. 14). Rather, Richiboucto Harbour is predicted to have greater seston depletion in response to SBOA present in Aldouane (Fig 14-A) and Bedec (Fig 14-B), but the amount and locale of depletion is not uniform. Similarly, the comparison of SDI scores between Fig 14-B and C illustrates that there is greater phytoplankton depletion in Bedec as a result of SBOA sites in Richiboucto Harbour. These results suggest that oyster filtration services can benefit adjacent bays, although the benefits are contextualized to the specific bay and its SBOA conditions. The SBOA sites and resulting ZOI within Bedec was further examined given the bay's isolated nature. The numerical model had estimated the area of Bedec to be 4.84km^2 , of which TAL area was estimated to be 18.7%⁴¹ while the ZOI due to phytoplankton depletion ($\text{SDI} > 1$) was 58% of the bay area. Therefore, the ZOI in Bedec is >3 times larger than the area of oysters, suggesting that the area of phytoplankton depletion can have a significant and widespread effect on the bay.

⁴¹ In comparison, the EC/DFO ground-truth data indicates that TAL area for Bedec is 14% of total bay area. This discrepancy can be explained by the difference in methodologies used to calculate TAL area as well as the hydrological chart differences to calculate Bedec's boundaries and therefore area.

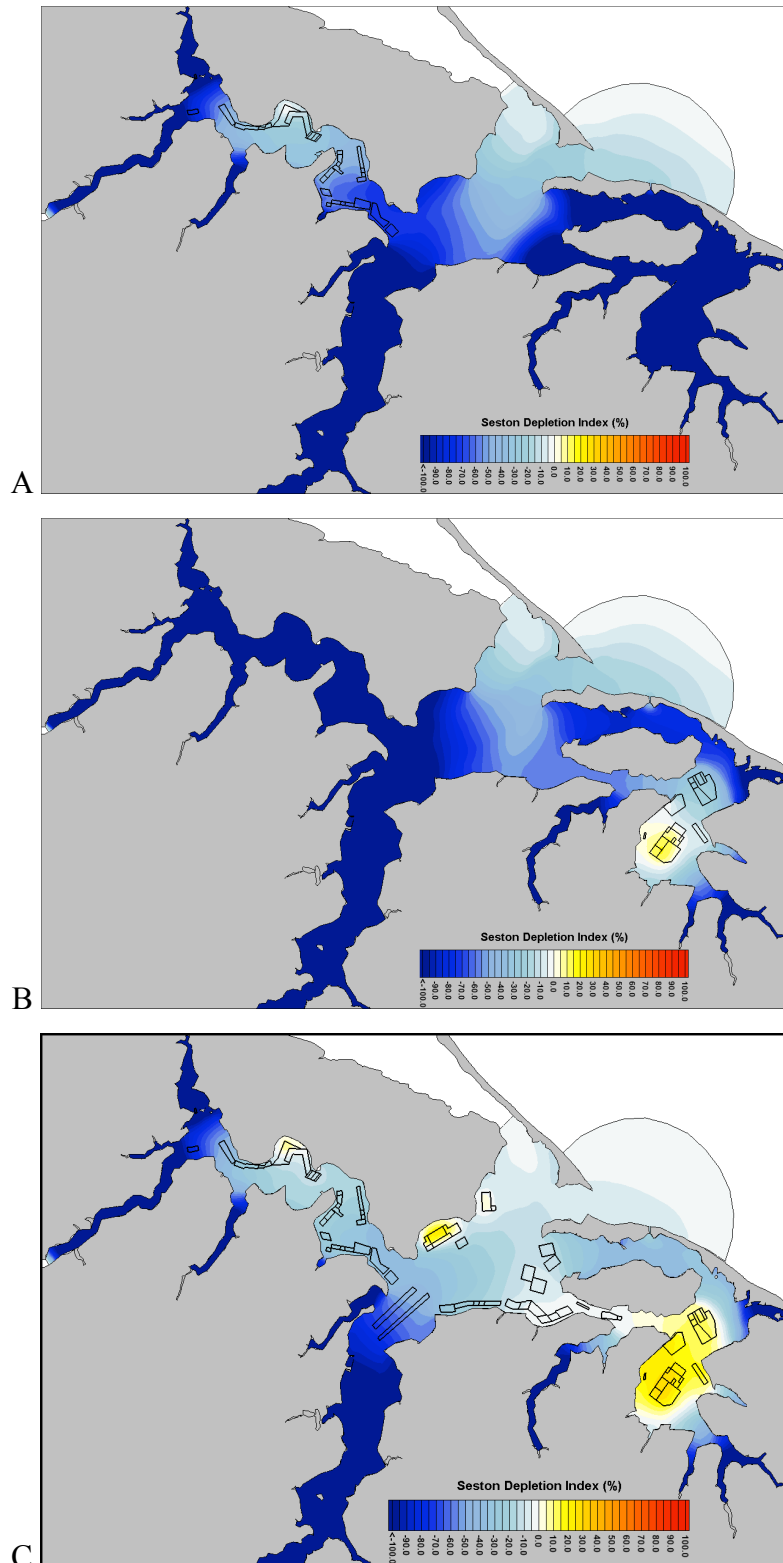


Figure 13. Digital areal images illustrating the seston depletion index of the Richiboucto estuary in response to different SBOA conditions produced by the numerical model (Guyondet et al., 2013). A) with SBOA present in Aldouane only; B) with SBOA present in Bedec only; C) with SBOA leases throughout the Richiboucto estuary (real scenario).

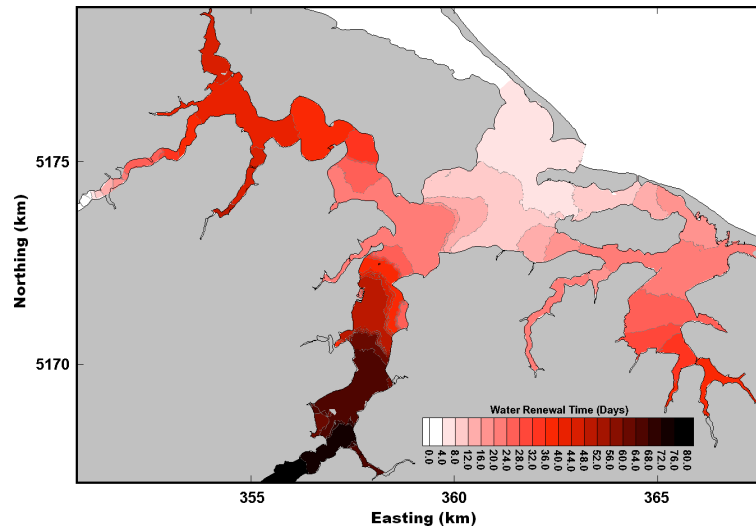


Figure 14. The water renewal time for areas of the Richiboucto estuary measured in days. Adapted from Guyondet et al., 2013.

8.4.2. Bay-Scale Statistical Analyses

8.4.2.1. Linear & Multiple Regression Analysis

The linear regressions found that at the bay-scale, only the bay area co-variable was significantly correlated to eelgrass cover ($P=0.00845$; adj. $R^2=0.406$); in turn, all of the $n=5$ interaction terms were also found to have significant correlations with eelgrass cover. The MRA model yielded that all $n = 11$ bay-scale SBOA raw variables and interaction terms were significantly related to eelgrass cover ($P = 9.58e-07$ for $n = 1$ variable and $P = < 2e-16$ for $n = 10$ variables), disabling the model from dropping any variables from its data-frame (see Table 16 below). The highest correlation between the eelgrass data and the MRA model ($R^2 = 0.99$) was achieved when the data-frame was log transformed with the “Poisson” family (AIC = 184.48). The expected normal distribution of both the residuals against fitted values (RAFV) plot and the normal Q-Q plot further illustrates the significance of the MRA model (Fig. 16 and 17 below). The MRA revealed that TAL, bay area, gear, ASL: bay area, TSL: bay area, and bags: bay area were positively lending to eelgrass cover whereas ASL, TSL, bags, TAL: bay area, and gear: bay area had a negative influence on eelgrass cover.

The bay-scale linear regressions had yielded that only the co-variables associated with bay area (i.e., bay area [$P=0.008454$] and all interaction terms [$P=0.01737 < x < 0.04422$]) were statistically significant to determining eelgrass cover.

Contrastingly, none of the estuary-scale co-variables were found to be statistically significant with eelgrass cover. It is also worth noting that amongst the bay-scale linear regressions, gear area was found to be marginally statistically insignificant ($P=0.07754$). It is possible that with improved data accuracy gear area may have been found to be statistically significant; not only does gear area represent very small proportion of the total bay area (mean 0.03% of bay area), but simplified calculations were used to generate a rough estimate of the gear area.

Table 17. Summary table of the linear and multiple step-wise regression results at the bay-scale using the $n=11$ bay-scale raw and interacting SBOA co-variables. Significant results are indicated with bold text.

Co-Variable	Linear Regression Analysis		Multiple Regression Analysis	
	P value	Adjusted R ²	Estimate	P value
ASL	0.4313	-0.0266	-2.028e-01	< 2e-16
TSL	0.7447	-0.0734	-8.543e-02	< 2e-16
TAL	0.2924	0.01611	1.041e-01	< 2e-16
Bay area	0.008454	0.4061	4.786e-04	9.58e-07
Bag counts	0.3534	-0.005255	1.204e+01	< 2e-16
Gear area	0.07754	0.1734	-1.725e-04	< 2e-16
ASL: bay area	0.02619	0.4637	8.499e-05	< 2e-16
TSL: bay area	0.03601	0.4262	4.630e-05	< 2e-16
TAL: bay area	0.03425	0.4323	-4.911e-05	< 2e-16
Bags: bay area	0.01737	0.508	9.890e-08	< 2e-16
Gear: bay area	0.04422	0.4006	-7.077e-03	< 2e-16

Although there are three significant outliers observed within the RAFV plot (Richiboucto Harbour, St Simon Inlet, and St Simon North), the majority of the data is horizontally distributed along the residuals = 0 line and is thus indicating the linear relationship of eelgrass data and the MRA model, and that the variances of the error terms are equal (pers. comm. Joey Hartling June 17, 2013) (Fig. 15). Similarly, three outliers were observed within the normal Q-Q plots (St Simon Inlet, Tracadie South, and Tabusintac), yet the majority of the data was found diagonally distributed along the residuals = 0 line ($y = x$) which indicated that the eelgrass data and the MRA model are significantly similar to one another (pers. comm. Joey Hartling June 17, 2013) (Fig. 16).

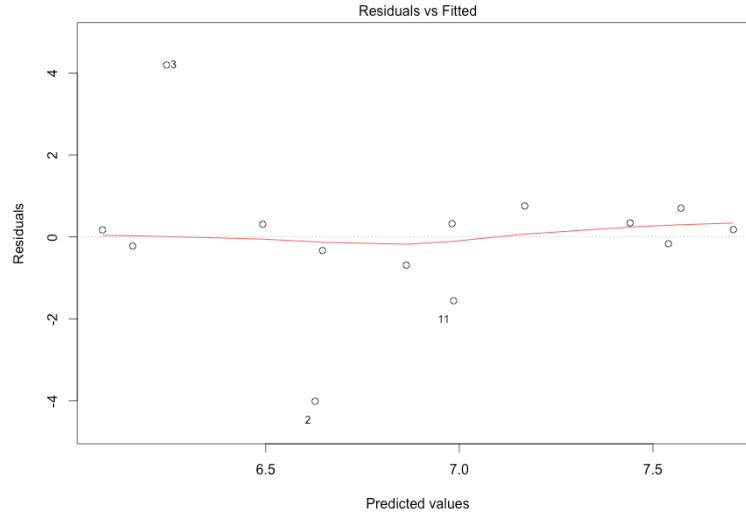


Figure 15. A plot of residuals against fitted values from the bay-scale multiple regression analysis. Data was found distributed in a horizontal band around the residuals = 0 line, illustrating that the variances of the error terms were equal as well as the data having a relatively linear relationship. Bays 2 (St Simon Inlet), 3 (St Simon North), and 11 (Richiboucto Harbor) are outliers.

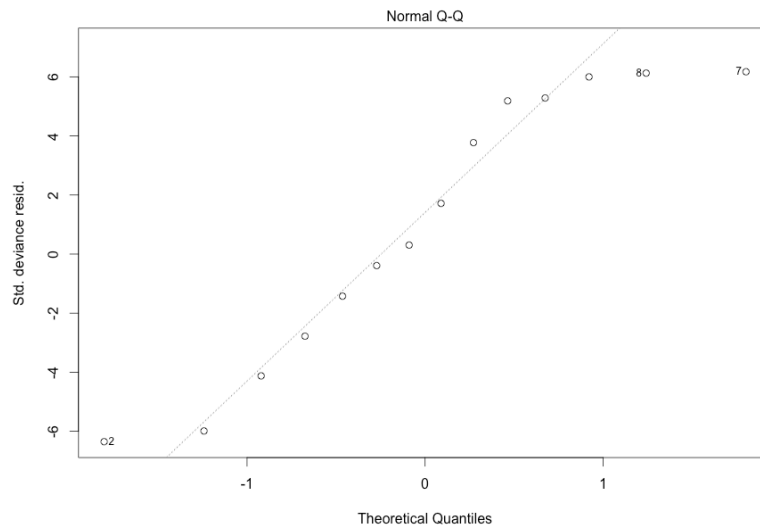


Figure 16. The Q-Q plots resulting from the bay-scale multiple regression analysis. The data was distributed along the residuals = 0 line ($y = x$), indicating that the eelgrass data and MRA model were significantly similar to one another. The deviation from the line at both tails of the plot indicate the presence of outliers at both extremes of the data; bays 2 (St Simon Inlet), 7 (Tracadie South), and 8 (Tabusintac) are outliers.

8.4.2.2. Principal Components Analysis

Because the raw independent variables were first transformed into z-scores and the mean of each variable became 0, a correlations matrix was used to generate the PCs. The PCA described the variance of data using all $n=6$ PCs (i.e., all $n=6$ co-variables were significant to the model), and thus yielded a six-dimensional space for which to

plot the PCs' eigen vectors. However, the majority of the variance is described in the first and second dimensions for a combined 78.8% of the variance explained.

Table 18. The percentage of variance explained by the n = 6 dimensions from the bay-scale Principal Components Analysis.

PCA Dimension	1	2	3	4	5	6
% Variance Explained	42.8	36.0	13.0	4.6	2.8	0.5

The variable factor map (VFM; Fig. 17) is a visual representation of the dimensions that best explain the PCA's rotation of the data, allowing the interpreter to observe relationships and patterns from the directions and associations of the eigen vectors. Based upon the general direction of the eigen vectors, it is evident that gear area, bag counts, and ASL are highly associated with one another while TSL and TAL are highly associated with a close association to bay area. These associations (i.e., gear/bags/ASL and TSL/TAL/bay area) are representative of their co-variables having positively correlated abundances for their given areas (e.g., the amount of bag counts, gear and ASL area are positively correlated), potentially indicative of a spatial segregation between these associations. For example, the amount of gear and bags present in a bay are spatially limited to ASL area.

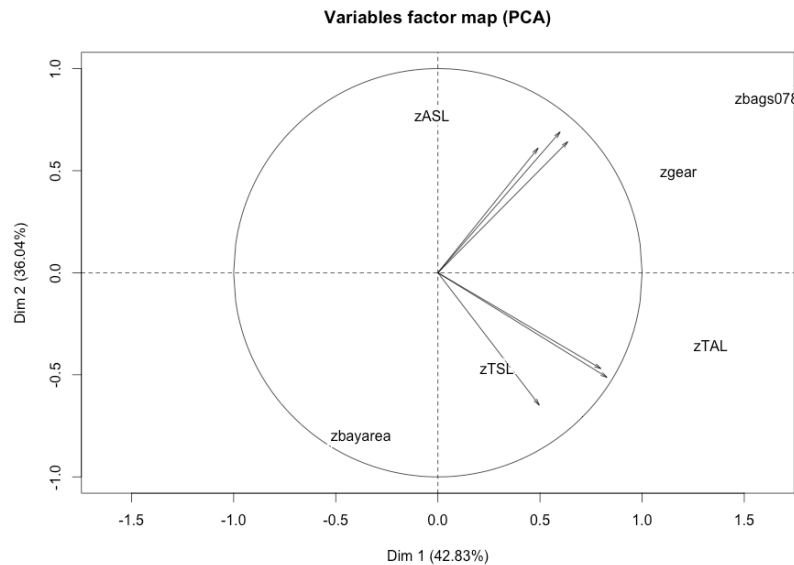


Figure 17. A variable factor map produced from a Principal Components analysis on the standardized bay-scale SBOA variables. The order of the co-variables' labels is in order of the eigen vectors, beginning with zASL and moving clock-wise. Dimension 1 explains 42.8% of the overall variance; dimension 2 explains 36.0% of the variance.

The correlations matrix and VFM results offer information as to which co-variables most heavily influence the variability of the data via their associations with the most significant PCA dimensions. Dimensions 1 and 2 are given the most consideration here since they explain the majority of the variance in the data. Both TAL and TSL area were found to be most closely associated to the first dimension (0.83 and 0.80 correlations, respectively) while gear area was found to be equally explained by both the first and second dimensions (0.64 correlation). Bags were primarily explained by the second dimension but were also well explained by the first dimension (0.69 and 0.60, respectively). Bay area was slightly more explained by the third dimension than the second (0.53 and 0.50 correlations, respectively) and ASL was least explained by the first dimension (0.49 correlation) but most explained by the second dimension (0.61 correlation). Because dimension 1 contributes the most variance explained to the PCA, and because TAL and TSL were the most explained by dimension 1, then TAL and TSL are therefore the co-variables that most influence the variability of the data at the bay scale.

8.4.2.3. Sensitivity Analysis

Using a sensitivity analysis (SA) to create theoretical minimum and maximum values for each of the $n = 5$ bay-scale co-variables examined (see Methods section 2.4.2.5 and Appendix 8.2.3.3. for a description of the data and model), the MRA model was used to predict eelgrass cover in response to different co-variable values for each of the $n=14$ bays. The SA allowed the interpreter to examine the amount of change in eelgrass cover due to increases and decreases of individual co-variables, in turn describing specific bay-scale interactions and relationships between eelgrass and SBOA. The SA found that compared to the actual EC/DFO data⁴², the model predicted eelgrass cover to be greatest in response to the minimum ASL, TAL, and bag count values and in response to maximum TSL values (% Difference in Table 19). There was no reported change in eelgrass cover in response to the manipulated gear area values. Interestingly, both minimum and maximum values for the TAL and bag count co-variables produced an increase in predicted eelgrass cover relative to the ground-truth data.

⁴² The SA results were compared to the mean EC/DFO eelgrass cover given that it is ground-truth data, as well as there being a negligible mean difference between it and the predicted MRA eelgrass cover per bay area values.

Table 19. The sensitivity analysis results at the bay-scale, representing each of the n = 5 manipulated SBOA values, expressed in both mean area (ha) and mean percent predicted eelgrass cover per bay (%) with n = 14 bays per analysis. Results are compared to mean EC/DFO eelgrass cover data to create the mean percent difference in eelgrass cover (% difference) in response to an increase or decrease in the SBOA values.

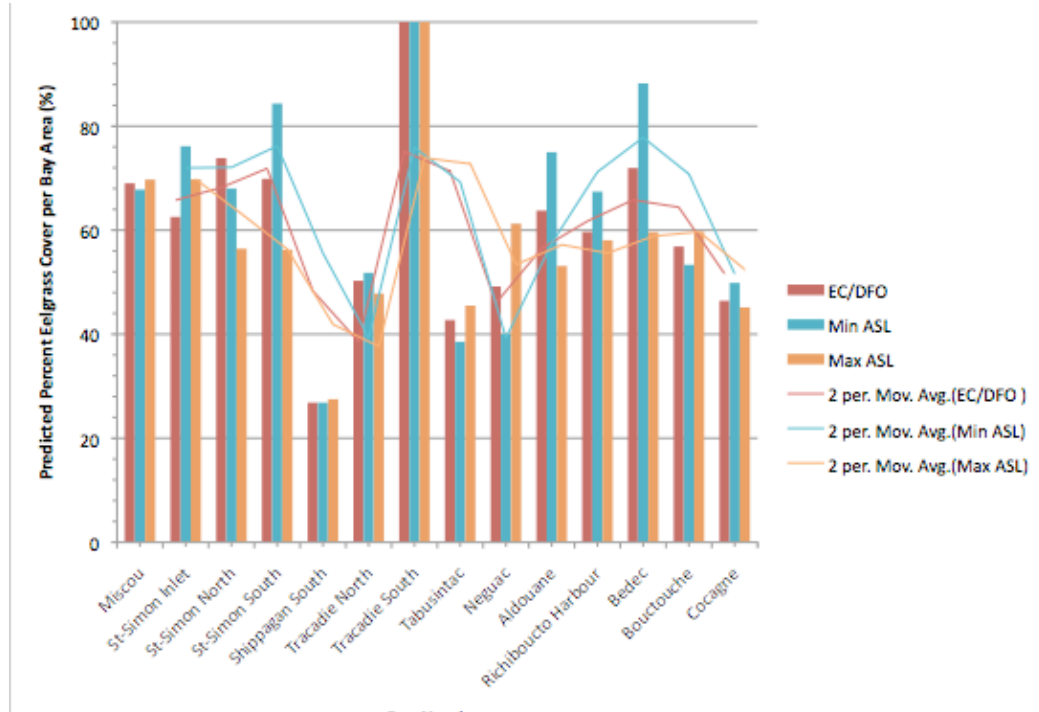
	<i>EC/DFO</i>	Min ASL	Max ASL	Min TSL	Max TSL	Min TAL	Max TAL	Min Bags	Max Bags	Min Gear	Max Gear
Mean Predicted Eelgrass Cover (ha)	1124.8	1127.4	1139.3	926.2	1676.5	2221.1	1000.8	1060.1	1267.6	1127.6	1126.4
Mean Predicted Eelgrass Cover per Mean Bay Area (%)	67.4	70.6	64.9	67.3	79.7	90.7	81.8	69.8	67.9	67.4	67.4
% Difference	/	+3.3	-2.5	-0.1	+12.3	+23.4	+14.4	+2.4	+0.5	0.0	0.0

Outliers were identified as the values that resulted in eelgrass cover predictions that were greater than the respective bay area. Other than for Tracadie South, which resulted in all n = 10 predicted eelgrass cover to be >100% of the bay area⁴³, the outliers were found exclusively within the TSL and TAL co-variable manipulations. The maximum TSL and minimum TAL values resulted in outlier eelgrass cover response for both Miscou (175.0% and 256.2% eelgrass cover per bay area, respectively) and Neguac (106.7% and 252.1%, respectively). Inversely, the SA results for St Simon South yielded eelgrass cover responses greater than its bay area in response to minimum TSL and maximum TAL values (121.5% and 229.7%, respectively). Outliers were additionally found in response to only maximum TAL values for St Simon Inlet (100.8%), St Simon North (109.1%), and Bedec (124%). These results are likely attributed to the PCA findings that TAL and TSL are responsible for the greatest variance in the data, and therefore have the greatest influence on the SA results; it was not possible for the SA model to correct for this disproportionately high variance explained. However, with the exception of Tracadie South results, only n=8 of the total n=140 SA results (n=14 bays*n=10 manipulations) were found to be outliers, suggesting

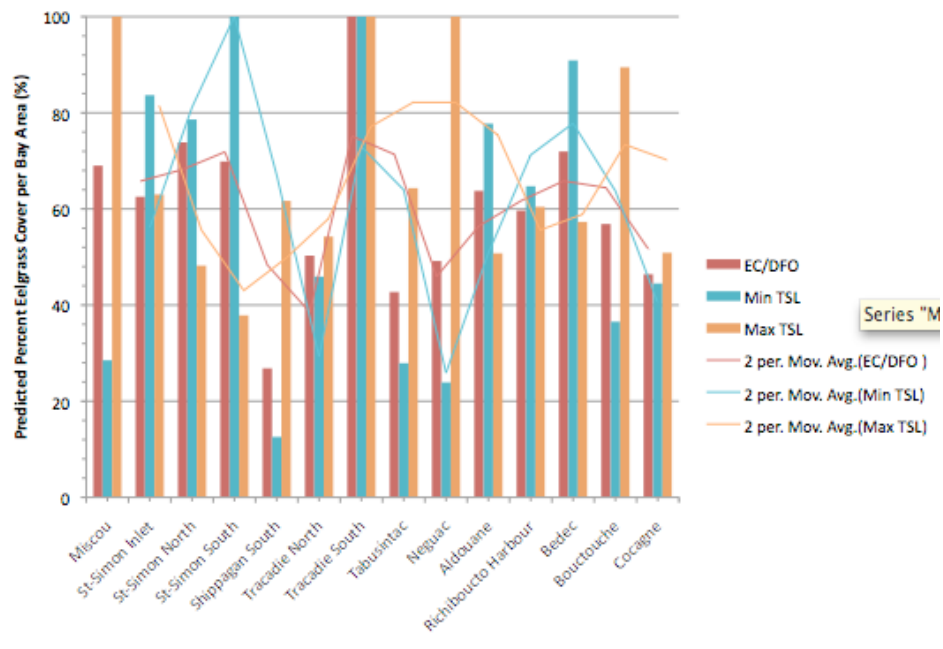
⁴³ It was not unexpected to have found that all n = 10 predictions made for Tracadie South were > 100% of the bay area given that the predicted eelgrass cover for the bay prior to co-variable manipulation was also >100% of bay area.

that despite these high variances explained the SA model was adequate to predicted eelgrass cover. To best interpret the results visually, these outliers were set at the realistic maximum eelgrass cover per bay (100%) to allow the interpreter to examine the scale-specific relationships and patterns between eelgrass cover and SBOA co-variables.

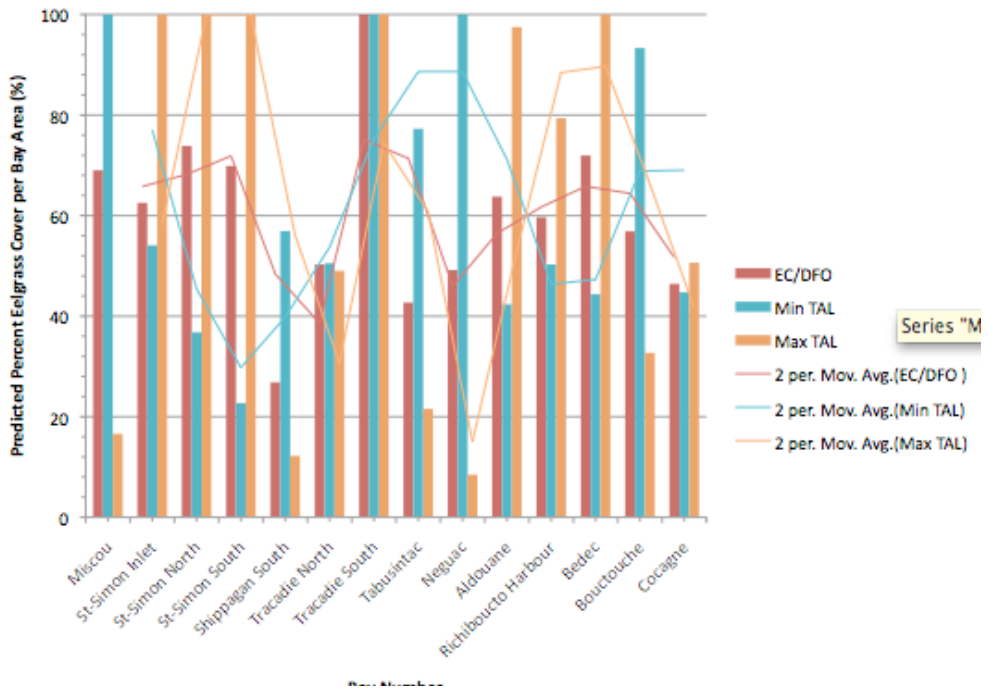
A



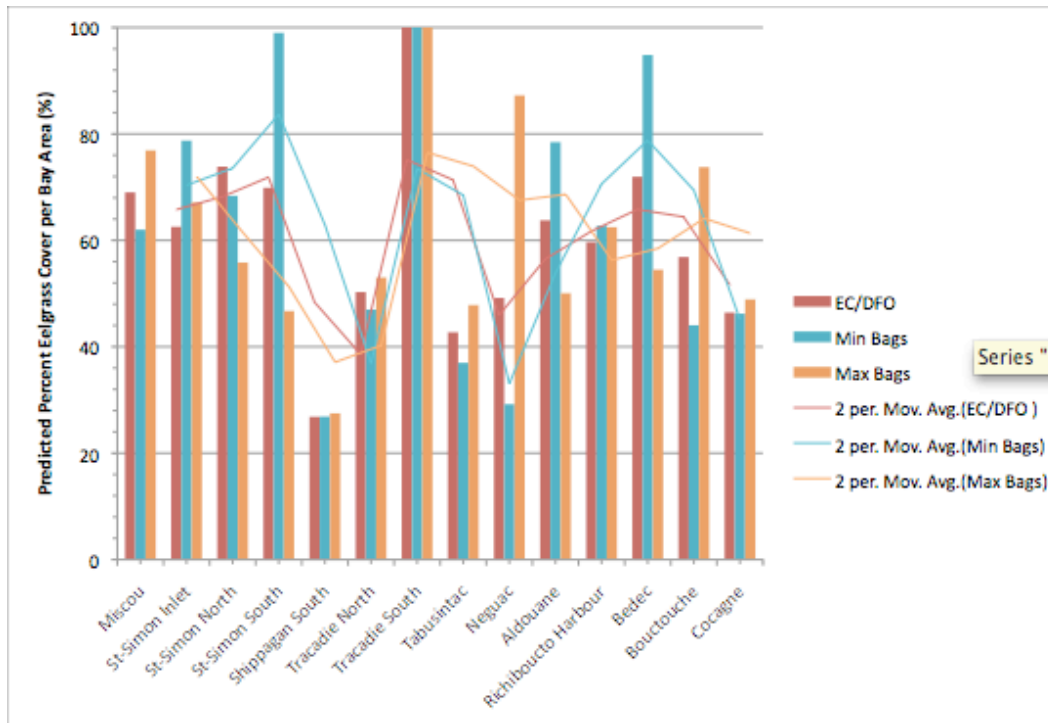
B



C



D



E

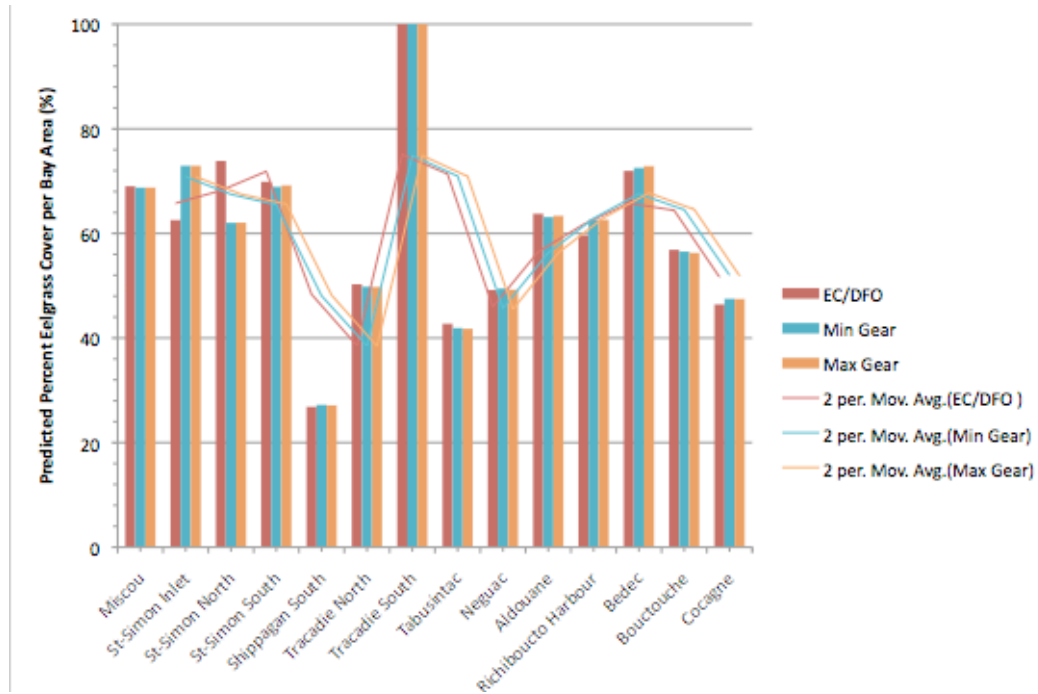


Figure 18. Histograms of the percent eelgrass cover per bay area (%) for each of the n = 14 bays as the sensitivity analysis responds to the manipulated SBOA co-variables. The ground-truth EC/DFO eelgrass cover is plotted alongside the predicted eelgrass cover results from the n=10 sensitivity analyses that examined the effects of minimum and maximum values for each of the n=5 SBOA co-variables: (A) ASL, (B) TSL, (C) TAL, (D) bag counts, (E) gear area.

The bay-scale SA results were analyzed and plotted, and the visual interpretation of the results was guided by including the lines of moving averages (LMAs) across bays for each manipulated co-variable. On average, the mean eelgrass cover response to minimum ASL was greater than to maximum ASL with a difference of 5.8%; however, the patterns of LMAs are similar to one another and with the exception of Bedec, there are not significant departures from the ground-truth data (Fig 18-A). The SA results suggest that most bays will slightly benefit from less ASL. Contrastingly, the SA model had predicted on average an increase in eelgrass cover in response to maximum TSL than to minimum TSL, with a difference of 12.4%, although the responses were significantly dissimilar depending on the bay and the manipulated value observed (Fig 18-B). Although the eelgrass response to minimum TSL generally followed the same pattern as that of the ground-truth data (mean -0.1% difference), the response to maximum TSL values caused an exaggerated and inverted eelgrass response. Illustrated by the inverse eelgrass response to the minimum and maximum TSL values across the

bays, Fig 18-B appears to indicate a critical threshold between the manipulated TSL values such that the responses are bay-specific with each bay responding either very positively or negatively to the increase or decrease of TSL. Similarly, the manipulated TAL values illustrate an inverse response in predicted eelgrass cover, although eelgrass was predicted to increase the most in response to minimum TAL than to maximum TAL (mean 9% difference) (Fig 18-C). However, the predicted eelgrass responses to TAL were dissimilar and bay-specific; some bays greatly benefited from an increase in eelgrass cover in response to minimum TAL while others in response to maximum TAL. The inconsistent and inverse eelgrass responses appear to indicate a contextualized bay-scale critical threshold among manipulated TAL values. Figure 19-D suggests a similar trend of bag counts to ASL (Fig 18-A) such that predicted eelgrass cover marginally increased in response to minimum bag counts than maximum bag counts (mean 1.9% difference). The manipulated bag counts' LMA patterns further suggest a moderate deviation from the LMA of the ground-truth data, in turn suggesting a moderate influence of the change in bags counts onto predicted eelgrass cover, although the results are again contextualized to the specific bay observed. Lastly, the SA results for the manipulated gear area values suggest a no influence onto predicted eelgrass cover, as the mean difference for both the minimum and maximum values was negligible (mean 0% difference). However, Figure 18-E demonstrates contextualized small changes in predicted eelgrass cover responses, with some bays marginally benefitting from minimum gear area while others from maximum gears.

It is worthwhile to note that the trends in predicted eelgrass response to TSL and TAL are inverted to one another per bay. For bays in which an increase in eelgrass is predicted in response to minimum TAL, there is also a predicted increase for maximum TSL (e.g., Neguac); inversely, bays that respond positively to maximum TAL are also predicted to have increased eelgrass in response to minimum TSL (e.g., St Simon South). For example, St Simon South has 70% eelgrass cover per bay area (EC/DFO data) but is predicted to increase to 121.5% in response to minimum TSL and to decline to 38% in response to maximum TSL. Inversely, the same bay is predicted to have eelgrass cover decrease to 23% bay area in response to minimum TAL but increased to 230% cover per bay in response to maximum TAL. In comparison to St Simon South,

Neguac has 49% eelgrass cover per bay area (EC/DFO data) but is predicted to decline to 23.9% in response to minimum TSL and to increase to 106.7% in response to maximum TSL. Contrastingly, Neguac's eelgrass cover is predicted to increase to 252% cover per bay with minimum TAL and decreasing to 8% in response to maximum TAL. These similar trends but inverted relationships suggest a significant spatial segregation between TSL and TAL, and that a critical threshold likely exists between them (e.g., for St Simon South, the critical threshold is between 226 ha of TSL and 259 ha of TAL). Moreover, the critical thresholds are contextualized to each bay such that every bay is responding dissimilarly to the manipulated TSL and TAL values. It is important recall that the minimum and maximum values of all co-variables, including TSL and TAL, vary across all n=14 bays since the normalized means were multiplied and divided by each bay's variables. Therefore, each bay was manipulated by dissimilar amounts, although these amounts are theoretically appropriate for each bay given that they are based upon normalized means for each co-variable.

The simultaneous increase in predicted eelgrass cover in response to both the minimum and maximum values of TSL and bag counts is indicative that the bays are responding contextually to the manipulated values. That is, all bays are responding positively to the manipulated TSL and bag count values, although the response is contextualized to the bay and manipulation. On average, bays are responding more positively to the minimum values than to the maximum values of TSL (mean +23.4 and +14.4%, respectively) and bag counts (+2.4% and +0.5%, respectively). These trends are reinforced upon observing the marginal difference in predicted eelgrass cover in response to the mean of all n=5 maximum values (72.3% eelgrass per bay) and the mean of all n=5 minimum values (73.2% eelgrass per bay), both of which are greater than the mean EC/DFO eelgrass cover per bay area (67.4%) (see Table 18, above). Therefore, the SA manipulation results are highly contextualized to the bay examined. It is uncertain why TSL and bag counts were the only two co-variables to have mean increased predicted eelgrass cover upon both manipulations.

8.4.3. Estuary-Scale Statistical Analyses

8.4.3.1. Linear & Multiple Step-Wise Regression Analyses

The linear regressions with the estuary and watershed-scale co-variables yielded no significant correlations to eelgrass cover. The multiple step-wise regression analysis (MRA) rid the gear area on the first drop, resulting in only n=5 estuary-scale SBOA co-variables being statistically significant to the model ($P < 2e-16$ for all n=5 co-variables). The highest correlation between the eelgrass data and the MRA model ($R^2 = 0.89$) was achieved when the data-frame was log transformed with the “Poisson” family (AIC = 246.85). The expected normal distribution of both the residuals against fitted values (RAFV) plot and the normal Q-Q plot further illustrates the significance of the MRA model (see Appendix 8.5.2.1. for results summary and plots). The MRA had indicated that ASL, TSL, estuary area had a positive influence on eelgrass cover while TAL, gear area, and bags had a negative influence.

Table 20. Summary table of the linear and multiple step-wise regression (MRA) results at the estuary scale using the n=6 estuary and n=6 watershed-scale SBOA co-variables. Significant results are indicated with bold text. Gear area was dropped from the MRA; the MRA was not preformed using the watershed co-variables due to high correlations.

Co-Variable	Linear Regression Analysis		Multiple Regression Analysis	
	P value	Adjusted R ²	Estimate	P value
ASL	0.196	0.09912	8.259e-03	< 2e-16
TSL	0.1853	0.109	1.493e-02	< 2e-16
TAL	0.1751	0.1188	-8.633e-03	< 2e-16
Estuary area	0.1627	0.1519	1.166e-03	< 2e-16
Bag counts	0.3326	0.006993	-2.007e-05	< 2e-16
Gear area	0.5388	-0.06987	dropped	dropped
AGR	0.8178	-0.1555	/	/
IND	0.6372	-0.1206	/	/
INF	0.7648	-0.1479	/	/
SET	0.8836	-0.1622	/	/
Forest	0.3362	0.01303	/	/
Watershed	0.3762	-0.01257	/	/

The expected normal distribution of both the residuals against fitted values (RAFV) plot and the normal Q-Q plot further illustrates the significance of the MRA model. Although there is one significant outlier observed within the RAFV plot (Tabusintac), the majority of the data is horizontally distributed along the residuals = 0 line and is thus indicating the linear relationship of eelgrass data and the MRA model. However, the data does not lie along the residuals = 0 line, meaning that the variances of

the error terms are not equal (pers. comm. Joey Hartling June 17, 2013) (Fig. 19). Two outliers were observed within the normal Q-Q plots (Tabusintac and Cocagne), yet the majority of the data was found diagonally distributed along the residuals = 0 line ($y = x$) which indicated that the eelgrass data and the MRA model are significantly similar to one another (pers. comm. Joey Hartling June 17, 2013) (Fig. 20).

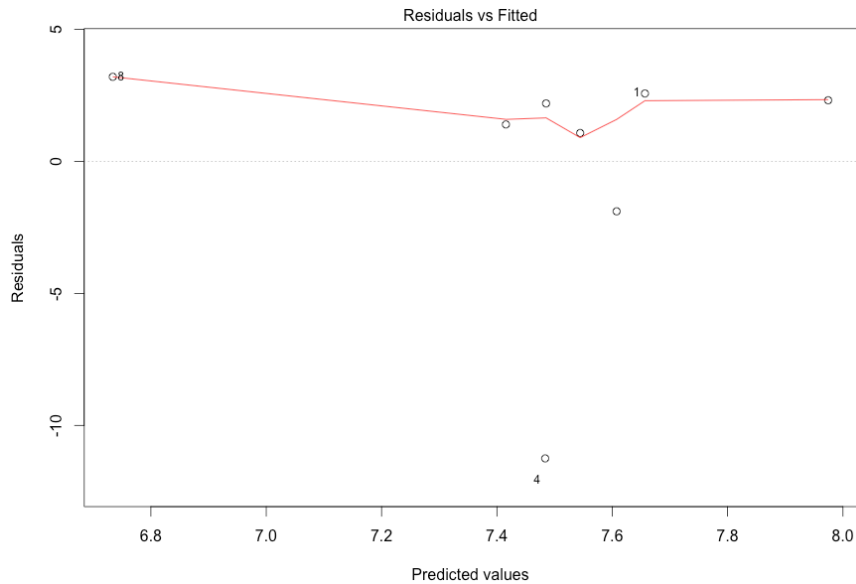


Figure 19. A plot of residuals against fitted values resulting from the estuary-scale multiple regression analysis. Data distribution was shaped as a horizontal band above the residuals = 0 line, illustrating a relatively linear relationship within the data although the variances of the errors terms were not found to be equal. Bay 4 (Tabusintac) is an outlier.

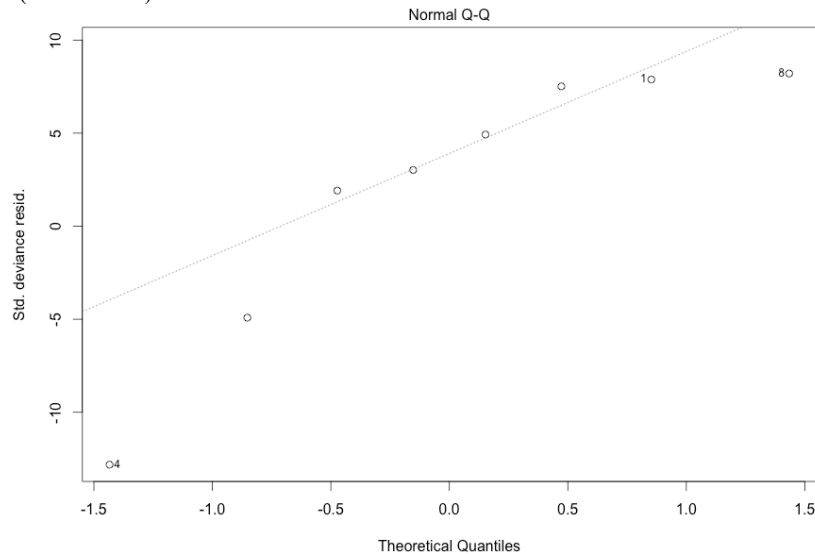


Figure 20. The Q-Q plots resulting from the estuary-scale multiple regression analysis. The data was found distributed along the residuals = 0 line ($y = x$), indicating that the eelgrass data and MRA model were significantly similar to one another. The deviation from the line at both tails of the plot indicate the presence of outliers at both extremes of the data; bays 4 (Tabusintac) and 8 (Cocagne) are outliers.

8.4.3.2. Principal Components Analysis

8.4.3.2.1. Estuary-Scale PCA

Similar to the bay-scale PCA, a correlations matrix was used to generate the estuary-scale PCs given that the data was first transformed into z-scores and the resulting means were 0. Additionally, gear was included in the PCA, unlike the MRA, given that the PCA is robust to correlations. The PCA described the variance of data using all $n = 6$ PCs; that is, all $n=6$ estuary-scale co-variables were significant to the model, resulting in the creation of $n=6$ dimensions. However, the PCA had yielded a five-dimensional space to plot the PCs' eigen vectors, meaning that one dimension was not required since it is well represented by another dimension that can similarly describe the variance explained. It is proposed that dimension 5 and 6 were combined to represent the fifth of $n=5$ plotted dimensions in the variable factor map (VFM; Fig. 23) given that both represent $<1.0\%$ of the variance explained (pers. comm. Stu Carson, October 9, 2013). The majority of the variance is described in the first dimension, representing 61.7% of the variance explained (Table 21).

Table 21. The percentage of variance explained by each of the $n = 6$ dimensions resulting from the estuary-scale Principal Components Analysis.

PC Dimension	1	2	3	4	5	6
% Variance Explained	61.7	19.6	16.0	1.6	0.9	0.3

The variable factor map (VFM; Fig. 21) is a visual representation of the dimensions that best explain the PCA's rotation of the data, allowing the interpreter to observe relationships and patterns from the directions and associations of the eigen vectors. Based upon the general direction of the eigen vectors, it is evident that gear area and ASL are most closely associated with one another while TSL and TAL are highly associated. The bag counts' eigen vector is almost perfectly correlated to the first dimension. The eigen vector for the estuary area is not especially associated with any variable, but is between the TAL/TSL and gear/ASL associations. These gear/ASL and TSL/TAL associations are representative of the SBOA co-variables having positively correlated abundances in the areas they appear in, potentially indicative of a spatial segregation between these interacting sets of co-variables. For example, both SBOA gear and bags are spatially limited to the ASL area under provincial regulation, while

TSL is a subset to TAL and both represent a mean of 6.0 and 7.9% of estuary area, respectively.

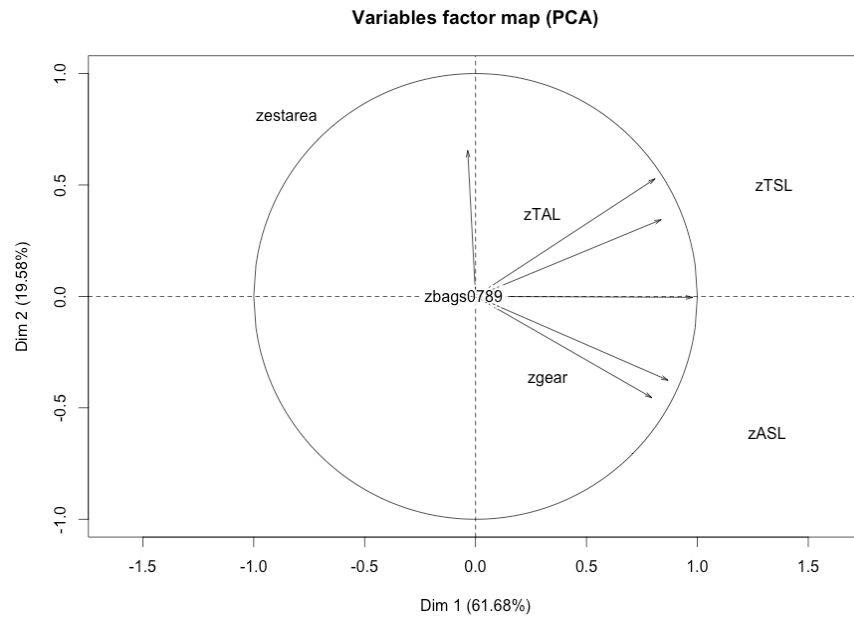


Figure 21. A variable factor map (VFM) produced from a Principal Components analysis on the standardized estuary-scale SBOA variables. The order of the co-variables' labels is in order of the eigen vectors, beginning with z-estuary area and moving clockwise. Dimension 1 explains 61.7% of the overall variance; dimension 2 explains 19.6% of the variance.

The correlations matrix and VFM results offer information as to which co-variables most heavily influence the variability of the data via their associations with the most significant PCA dimensions. Dimension 1 is given the most consideration here since it has explained the vast majority of the variance in the data, with some mention to dimensions 2 and 3. Bag counts were found to be most closely associated to the first dimension (0.98 correlation) with gear area being the second most closely associated co-variable to dimension 1 (0.87 correlation). The ASL, TSL, TAL areas were also found to be most highly correlated to dimension 1 (0.8 correlations each) compared to the other five dimensions. Only estuary area was found to have a negative association with dimension 1 (-0.04 correlation); instead, it was found most highly correlated with the third dimension (0.75 correlation). These results suggest that almost all of the co-variables' PCs lend significantly to dimension 1, and therefore significantly influence the variability of the data.

8.4.3.2.2. Watershed-Scale PCA

The watershed co-variables were similarly analyzed using the PCA. A correlations matrix was used to analyze the n=6 co-variables; resulted yielded that that all co-variables were significant to the model, resulting in the creation of n=6 dimensions. However, the PCA had yielded a five-dimensional space to plot the PCs' eigen vectors, meaning that one dimension was not required since it is well represented by another dimension that can similarly describe the variance explained. It is proposed that dimension 5 and 6 were combined to represent the fifth of n=5 plotted dimensions in the variable factor map (VFM; Fig. 22) given that both represent <1.0% of the variance explained (pers. comm. Stu Carson, October 9, 2013). The majority of the variance is described in the first dimension, representing 62.2% of the variance explained.

Table 22. The percentage of variance explained by each of the n = 6 dimensions resulting from the watershed-scale Principal Components analysis.

PC Dimension	1	2	3	4	5	6
% Variance Explained	62.2	20.7	13.2	3.0	1.0	<0.01

Based upon the variable factor map (VFM; Fig. 24), watershed-scale relationships and patterns become apparent from the directions and associations of the PCs' eigen vectors. Although the eigen vectors are difficult to interpret here given the mismatched labels, the results suggest that AGR, forest, and watershed area are closely associated with one another, and are the most closely correlated to dimension 1. The eigen vectors for INF, SET, and IND are not closely associated to any other variables. The ARG/forest/watershed correlations are representative that the watershed co-variables have positively correlated abundances in the areas they appear in, potentially indicative of a spatial segregation between these interacting sets of co-variables. For example, AGR is the largest non-forested land-use area (non-forested area is perfectly correlated yet inversely related to forested area) and the second largest contributor to watershed area after forested area.

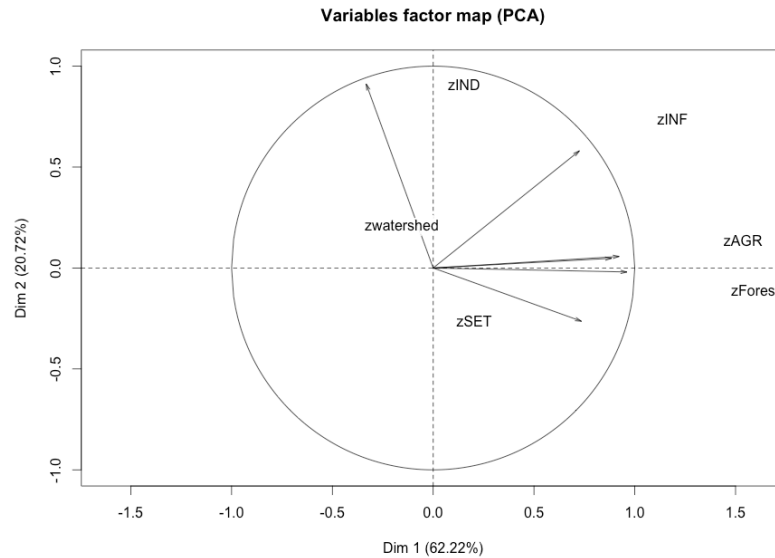


Figure 22. A variable factor map (VFM) produced from a Principal Components analysis on the standardized watershed-scale SBOA variables. The order of the co-variables' labels is in order of the eigen vectors, beginning with z-watershed area and moving clockwise. Dimension 1 explains 62.2% of the overall variance; dimension 2 explains 20.7% of the variance.

The correlations matrix and VFM results offer information as to which co-variables most heavily influence the variability of the data via their associations with the most significant PCA dimensions. Dimension 1 is given the most consideration here since it has explained the vast majority of the variance in the data, with some mention to dimensions 2 and 3. AGR was found to be most closely associated to the first dimension (0.96 correlation) with watershed area being the second most closely associated co-variable to dimension 1 (0.92 correlation). The forest, SET, and INF areas were also found to be most highly correlated to dimension 1 (0.88, 0.74, and 0.73 correlation, respectively) compared to the other five dimensions. Only IND was found to have a negative association with dimension 1 (-0.33 correlation); instead, it most highly correlated with the second dimension (0.91 correlation). These results suggest that almost all of the co-variables' PCs lend significantly to dimension 1, and therefore significantly influence the variability of the data.

8.4.3.3. Sensitivity analysis

Using a sensitivity analysis (SA) to create theoretical minimum and maximum values for each of the $n = 4$ estuary-scale co-variables examined⁴⁴, the MRA model was

⁴⁴ The SA is based upon the MRA model to make eelgrass cover predictions, and because the MRA had dropped gear area from its data-frame, the manipulated gear values were not included in the SA.

used to predict eelgrass cover response for each n=8 estuaries. The SA allowed the interpreter to examine the amount of change in eelgrass cover as the model responded to the individual increases and decreases of the SBOA values, in turn describing specific bay-scale interactions and relationships between eelgrass and SBOA. The SA found that compared to the EC/DFO data⁴⁵, the model had predicted the mean eelgrass cover to increase in response to the mean maximum ASL and TSL (+0.99% and 11.22%, respectively) values and mean minimum TAL and TAL and bag count (+10.25% and 0.74%, respectively) values (see % Difference in Table 23).

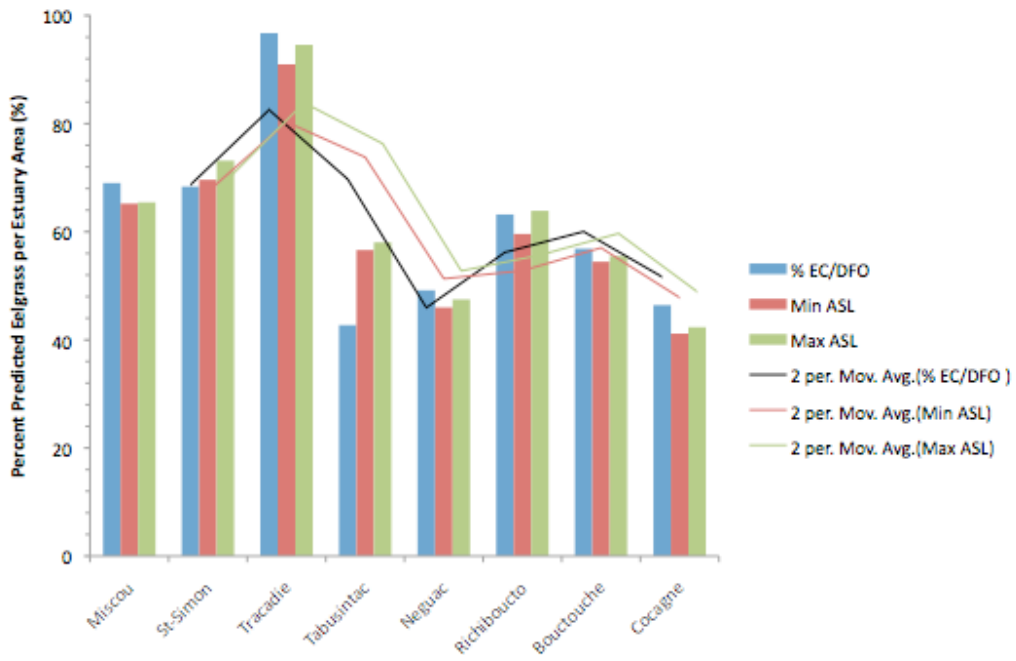
Table 23. The mean SA results for each of the n = 10 manipulated SBOA values, expressed in both area (ha) and percent predicted eelgrass cover per bay (%) with n = 14 bays per analysis. Results are compared to mean EC/DFO eelgrass cover data to create the mean percent difference in eelgrass cover (% difference), providing information as to the mean growth and/or decline in response to an increase or decrease in the SBOA values.

	EC/DFO	Min ASL	Max ASL	Min TSL	Max TSL	Min TAL	Max TAL	Min Bags	Max Bags
Mean Predicted Eelgrass Cover (ha)	1124.8	1127.4	1139.3	926.2	1676.5	2221.1	1000.8	1060.1	1267.6
Mean Predicted Eelgrass Cover per Mean Bay Area (%)	61.6	60.5	62.6	52.7	72.8	71.8	52.3	62.3	60.7
% Difference	/	-1.11	+0.99	-8.82	+11.22	+10.25	-9.27	+0.74	-0.91

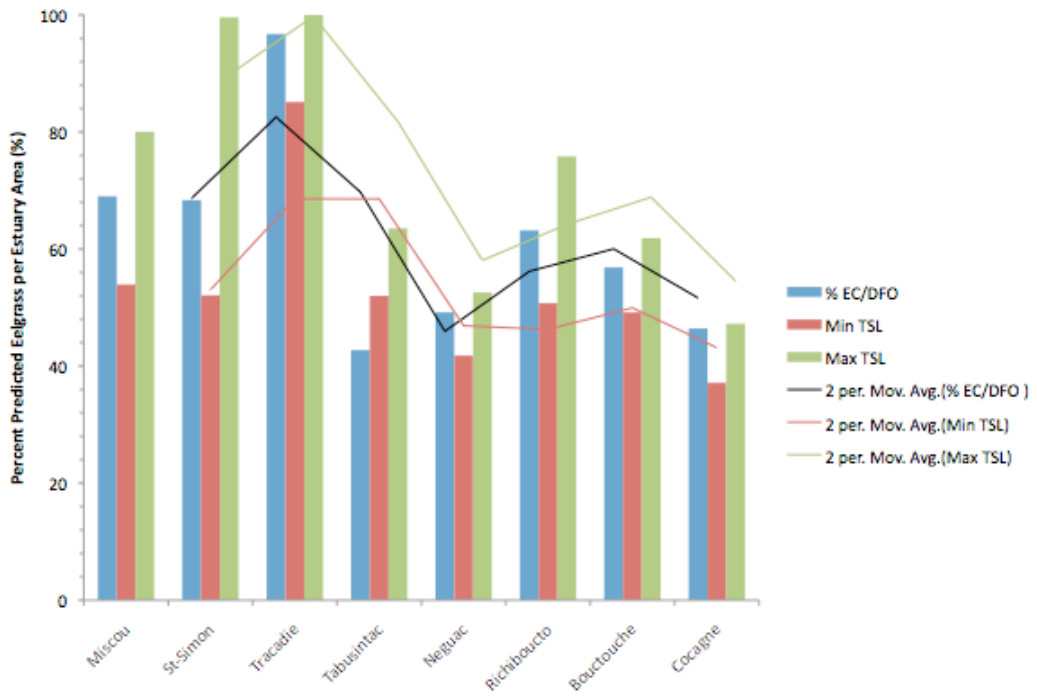
Only one outlier was identified to have a predicted eelgrass cover greater than the respective estuary area (i.e., >100% eelgrass cover per estuary area); Tracadie was found to have 101.5% eelgrass cover relative to its estuary area in response to maximum TSL. This outlier was set to the realistic maximum of 100% for better visual interpretation in order to examine the scale-specific relationships and patterns between eelgrass cover and SBOA co-variables (Fig. 23).

⁴⁵ The SA results were compared to the mean EC/DFO eelgrass cover given that it is ground-truth data, as well as there being a negligible mean difference between it and the predicted MRA eelgrass cover per bay area values.

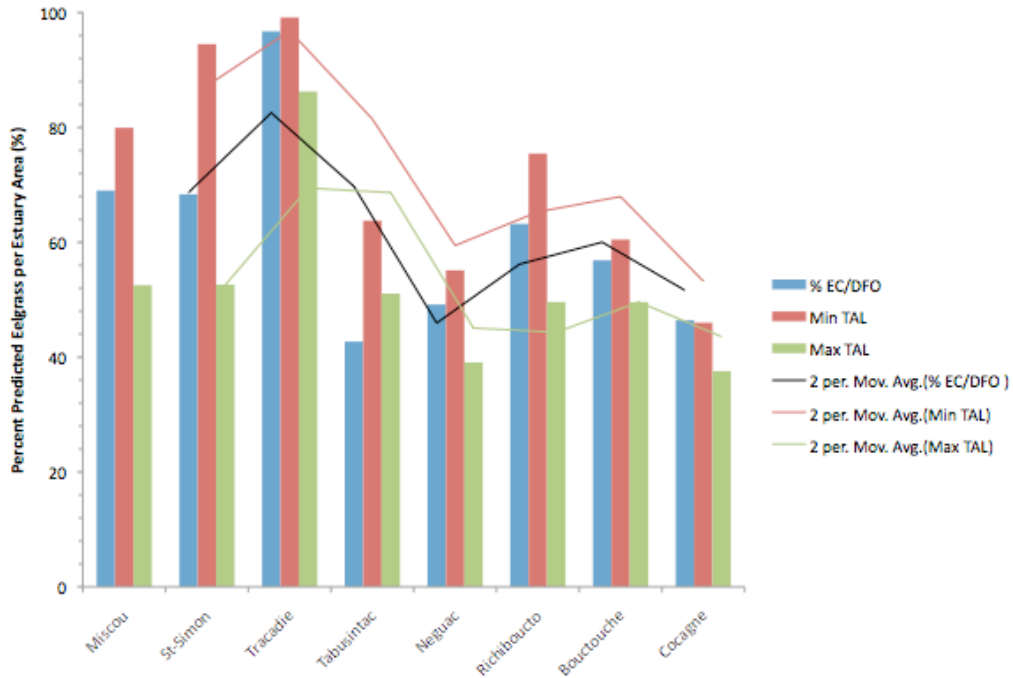
A



B



C



D

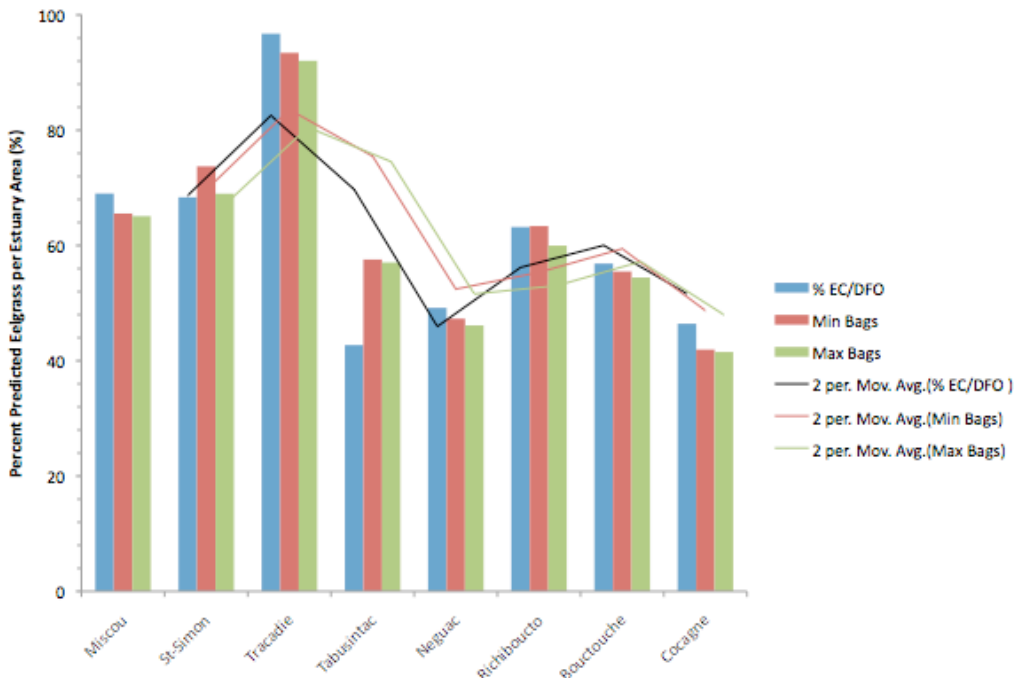


Figure 23. Histograms of the percent eelgrass cover per estuary area (%) for each of the n = 8 estuaries as the sensitivity analysis responds to the manipulated SBOA co-variables. The ground-truth EC/DFO eelgrass cover is plotted alongside the predicted eelgrass cover results from the n=8 sensitivity analyses that examined the effects of minimum and maximum values for each of the n=4 SBOA co-variables: (A) ASL, (B) TSL, (C) TAL, (D) bag counts.

The estuary-scale SA results were analyzed and plotted, and the visual interpretation of the results was guided by including the lines of moving averages (LMAs) across estuary for each manipulated co-variable. The manipulated ASL values had yielded greater predicted eelgrass cover in response to maximum ASL than minimum ASL (mean 2.1% difference), although the patterns of LMAs are somewhat similar to one another and with the exception of Tabusintac, there are not significant departures from the ground-truth data (Fig 23-A). The manipulated TSL results yield that the estuaries unanimously predict for greater eelgrass cover in response to maximum TSL compared to both ground-truth data as well as the minimum TSL results (mean 20.04% difference; Fig 23-B). However, the LMA patterns for the manipulated TSL values all appear to correspond with that of the ground-truth data, although grossly exaggerated, suggesting that the eelgrass-TSL relationship is consistent and not inversely related. Inversely, minimum TAL values were found to have the greatest predicted eelgrass cover response compared to the ground-truth and the maximum TAL values (mean 19.52% difference). Interestingly, the TAL values' LMAs were equal but opposite to those of the TSL values' LMAs (Fig 23-C), therefore suggesting that the eelgrass-TAL relationship is also consistent but opposite to that of the eelgrass-TSL relationship. Lastly, the LMAs for both the minimum and maximum bag counts revealed a pattern very similar to that of the ground-truth data (Fig 23-D) as well as to Figure 23-A. On average, eelgrass cover was predicted to marginally increase in response to minimum bag count values compared to ground-truth data, although the percent difference between the manipulated bag count values is almost negligible (mean 1.65% difference). Given the estuary-scale PCA results, it was expected that the largest deviance from the ground-truth data would be in response to the bag count values, although the SA results suggest a relatively small influence of bag counts onto predicted eelgrass cover compared to responses resulting from manipulated TSL and TAL values.

Interestingly, the co-variables' LMAs suggest that for many estuaries, the both the manipulate values predict eelgrass cover that is greater or lesser than the ground-truth data for any given estuary. It is hypothesized that this is occurring due to the fact that several of the estuaries are comprised of >1 bays, and so while the estuary-scale analyses only takes into account their mean values, it cannot account for the dissimilar

SBOA characteristics present in each of its contributory bays. Moreover, the same inverted relationship between TSL and TAL that was observed at the bays-scale persists at the estuary-scale. Again, the similar trends but inverted relationships suggest a significant spatial segregation between TSL and TAL, and that a critical threshold likely exists between them. However, the consistent LMA patterns suggest that the critical thresholds are a function of the TSL/TAL relationship, such that the eelgrass cover is highly predictable in response to the co-variable examined. Although these results are not expected from the estuary-scale PCA results, they are representative of the bay-scale PCA results. This result is somewhat expected considering that the examined estuaries are comprised of the same bays, and therefore the bay-scale results should still be represented in the estuary-scale results.

Moreover, there is evidence of relatively equal influences from all co-variables in the marginal difference in predicted eelgrass cover in response to the mean of all $n = 4$ maximum values (61.8%) and the mean of all $n = 4$ minimum values (62.0%), both of which are approximately equal to the mean EC/DFO eelgrass cover per bay area (61.6%). Therefore, the SA results and Fig 25 suggests that eelgrass cover responses are not estuary-specific; rather, they are proportionately uniform across estuaries, and that the estuaries respond to the manipulated SBOA co-variables in a predictive fashion.

8.4.4. Summary of Statistical Results

Consistent with the expected weight of effects at different spatial scales (Anderson et al., 2006), the statistical analyses found that the smaller bay-scale described greater significance and higher correlations than at the estuary-scale. The bay-scale linear regressions had yielded that only the co-variables associated with bay were statistically significant to determining eelgrass cover; contrastingly, none of the estuary-scale co-variables were found to be statistically significant with eelgrass cover. It was also found that bay-scale gear area was found to be marginally statistically insignificant however, the area and effect of gear varies significantly between different gear types (see McKindsey et al., 2006; Cranford et al., 2006). These differences would not have been representative in the small changes in gear area relative to bay area; as a result, the negative effects of higher oyster stoking densities (e.g., oyster cages) may have been negated by the low-impact of lower stocking density gears (e.g., long lines). Moreover,

the effect of the gear itself may be marginal in comparison to its significant ecological effect of habitat provision for fouling organisms that can increase pressures onto surrounding eelgrass, thus contributing to the negative near-field scale effects. These analyses did not take into consideration the fouling organisms and their additional effects, but gear area could be an inexact proxy for such effects; hence why the effect was a marginally statistically insignificant when the other raw co-variables (i.e., ASL, TAL, TSL, bags) were highly insignificant. Thus, gear area appears to have a disproportionately stronger effect on eelgrass cover at the bay-scale, although it may simply be the cumulative impact of several SBOA sites contributing their respective near-field scale impacts of gear (Skinner et al., 2013) within every bay. Gear may therefore be an imperfect representation of the larger processes contributing to the shading, organic enrichment, nutrient alteration, or some combination thereof, that are known to cause eelgrass declines (Skinner et al., 2013).

The MRA results found that there was a greater correlation of eelgrass cover to the bay-scale model than to the estuary-scale model, although both models suggest very high correlations to eelgrass cover at their respective scales. Interestingly, the MRAs suggested that some co-variables offered differing contributions to eelgrass cover depending on the scale of the model. That is, while TAL and gear area were modeled to have positive contributions to eelgrass cover at the bay-scale, these co-variables were found to have negative contributions at the estuary-scale (note: the MRA had dropped gear area and is therefore not a true representative of the estuary-scale results). Inversely, ASL and TSL were found to have negative contributions to eelgrass cover at the bay-scale but positive contributions at the estuary-scale. It is interesting to note that the percent ASL, TSL, and TAL area per bay and estuary area do not significantly change between spatial scales (i.e., ASL: 2.4 and 2.6%; TSL: 6.4 and 6.0%; TAL: 8.1 and 7.9% per bay and estuary area, respectively). More research is required to determine the mechanisms driving these opposing results, although it is possible that each embayment is responding dissimilarly due to their unique processes and SBOA characteristics, as suggested by the literature (e.g., Anderson et al., 2006).

In both models, the bay and estuary areas positively contributed to eelgrass cover while bags offered a negative contribution. Of course, the larger the embayment

the greater the possible area for eelgrass to grow, and where the model was not given a spatial context, this result is entirely expected. The inverse relationship between bag count and eelgrass cover, however, suggests a constant negative relationship between the number of cultured oysters and eelgrass cover across both spatial scales⁴⁶. However, bays had the greatest and smallest estimated contributions to the bay- and estuary-scale MRA models, respectively (see Table 16 and 20), suggesting that the impact of the number of bags is much more pronounced at the bay than estuary scale.

Similar to the expected scale-dependent effects described by Anderson and colleagues (2006), the bay-scale PCA yielded distinctive results compared to the estuary- and watershed-scales that appear to be too large for eelgrass effects to resonate with the analyses. At the bay-scale, the PCA found that TSL and TAL were most responsible for the variance explained of eelgrass cover. Interestingly, TAL was found to have a slightly larger relative contribution to the variance explained than TSL (0.83 and 0.80, respectively) with the only difference between the two co-variables being that TAL accounts for the small amount of bottom lease culture remaining in the NB aquaculture industry. Although bottom oyster aquaculture practices and lease area are declining in NB, it would appear to have a significant contribution to eelgrass cover than in comparison to its absence. As suggested by the ongoing research by Courtenay and colleagues (2012), bottom table cultures can cause eelgrass die-off in a third of the time than SBOA (i.e., 21 versus 67 days, respectively) and would suggest that despite bottom culture not being a common method of oyster aquaculture in NB (mean 1.7% of bay area) it is a significant determinant of eelgrass cover in bays. The greatest change in predicted eelgrass cover was found in response to manipulated TSL and TAL values.

Upon increasing TSL area, it was found that eelgrass cover was predicted to increase at both the bay- and estuary-scales (+12.3 and +11.22%, respectively).

⁴⁶ However, it is important to note that the MRA is sensitive to the amount of data included in the model; these results may be indicative of a bias within the analysis such that the estuary-scale analyses had fewer co-variables to test (n=6) than the bay-scale (n=11) given the lack of degrees of freedom sufficient to include interaction terms in the estuary-scale MRA, as well as fewer number of estuaries (n=8) than bays (n=14). Despite the possible statistical biases, these results may be representative of the spatial scale segregation of effects; Anderson and colleagues (2006) suggest that the smaller the spatial scale the more pronounced the habitat effects, which thus far seem to be in accordance with the aforementioned results given that the ratio of SBOA co-variables to bay area is smaller than to estuary area and therefore the results were more significant.

Inversely, the increase of TAL was found to predict for a decrease in eelgrass cover at both bay- (-9% relative decrease; from 14.4 to 23.4%) and estuary-scale (-9.27%). Not only are these results inversely related to one another, but they vary intensely within spatial scales. That is, each bay and estuary has responded dissimilarly to the manipulated co-variables, unlike the results observed with the manipulated bag counts and ASL area that generally followed the same pattern as the ground-truth data. This suggests that both TAL and TSL highly influence predicted eelgrass cover and that there is a spatially explicit threshold between the two co-variables. On average, there are very small differences between the percent TSL and TAL area per bay area (6.4 and 8.1%, respectively) and per estuary area (6.0 and 7.9%, respectively), suggesting that the spatial thresholds at which the eelgrass cover is predicted to have opposing responses exist within these 1.7 and 1.9% differences of bay and estuary scales, respectively. Therefore, there appears to be an inflection point at which the ECC limits are reached and eelgrass cover is predicted to decline in response either an increase in either TAL or TSL, but that these thresholds are contextualized and highly bay- and estuary-specific.

It is also important to note that TAL and TSL have greater varying influences onto predicted eelgrass cover at the bay-scale than at the estuary scale, which is evident from the pattern of the LMAs. At the estuary-scale, the LMA patterns for both minimum and maximum TAL and TSL values appear to follow the general trend outlined in the ground-truth data; at the bay-scale, the minimum and maximum values are inversely related both within and between the TAL and TSL manipulations. The greater observed variance among eelgrass cover in response to manipulations suggests that the ecosystem is more sensitive to bay-scale effects than estuary-scale effects, as suggested by Anderson and colleagues (2006). The apparent effects of TAL and TSL at the bay-scale imply that both suspended bag and bottom table aquaculture practices can result in declining eelgrass cover; it is hypothesized that TAL exerted a greater affect than ASL because aquaculturalists will over-winter their bags throughout their total lease area and do not necessarily do so in the area that is actively used to culture the oysters. This can result in an affected area greater than the active aquaculture zone (i.e., ASL), that when combined with the known effects of bottom table cultures, can create significant negative effects. It

is hypothesized that the unique hydrodynamic and oceanographic processes present within each bay, that may otherwise be negate at the estuary-scale, may be responsible for the highly contextualized nature of these results. Wagner and colleagues (2012) similarly concluded that the observed eelgrass declines were largely dependent on the embayment's characteristics and the cultured oyster stocking densities.

9. Appendix B

9.1. R-Studio Code for Bay-Scale Statistics

9.1.1. Loading Data

```
eelgrass <- bays.raw.07.09.w.predict
eelgrass0789 <-as.numeric(eelgrass$X2007.2009.EC.DFO.eelgrass..ha.)
ASL <-as.numeric(eelgrass$active.suspension.lease..ha..2011)
TSL <-as.numeric(eelgrass$total.suspended.lease.area)
TBL <- as.numeric (eelgrass$X2011.bottom.lease..ha.)
TAL <-as.numeric(eelgrass$X2011.total.lease..ha.)
bayarea <-as.numeric(eelgrass$Bay.Area..ha.)
bags0789 <-as.numeric (eelgrass$X2007.2009.bags)
gear <-as.numeric(eelgrass$Gear.area..ha..2011)
```

9.1.2. Multiple Regression Analysis

```
newbaydata1 <-
data.frame(cbind(ASL,TSL,TAL,bayarea,bags0789,gear,EGmaxdepth,Ntissue,maxbaydepth)) ##bay scale
data frame
cor(newbaydata1) ##checking correlation in data
mod1 <- glm(eelgrass0789 ~
ASL+TSL+TAL+bayarea+gear+bags0789+ASL*bayarea+TSL*bayarea+TAL*bayarea+gear*bayarea+bag
s0789*bayarea,data=newbaydata1,family="poisson")
summary(mod1)
step1 <-step(mod1)
summary(step1) #####same, no drop
plot(mod1)
```

9.1.3. Eelgrass Predictions

```
eelgrass0789 <- rep(0,14)
predicteelgrass <-data.frame(cbind(eelgrass0789,ASL,TSL,TAL,bayarea,bags0789,gear))
predeelgrass <- predict(mod1,newdata=predicteelgrass,type="response")
predeelgrass
```

9.1.4. Principal Components Analysis

9.1.4.1. Z-Score Transformations

```
mean(eelgrass0789)
var(eelgrass0789)
zeelgrass <- (eelgrass0789-mean(eelgrass0789))/sqrt(var(eelgrass0789))
summary(zeelgrass)
plot(zeelgrass)
```

```
mean(ASL)
var(ASL)
zASL <- (ASL-mean(ASL))/sqrt(var(ASL))
summary(zASL)
plot(zASL)
```

```
mean (TSL)
var(TSL)
zTSL <- (TSL-mean(TSL))/sqrt(var(TSL))
summary(zTSL)
plot(zTSL)
```

```

mean (TAL)
var(TAL)
zTAL <- (TAL-mean(TAL))/sqrt(var(TAL))
summary(zTAL)
plot(zTAL)

mean (bags0789)
var(bags0789)
zbags0789 <- (bags0789-mean(bags0789))/sqrt(var(bags0789))
summary(zbags0789)
plot(zbags0789)

mean (bayarea)
var(bayarea)
zbayarea <- (bayarea-mean(bayarea))/sqrt(var(bayarea))
summary(zbayarea)
plot(zbayarea)

mean (gear)
var(gear)
zgear <- (gear-mean(gear))/sqrt(var(gear))
summary(zgear)
plot(zgear)

zscores <-data.frame (cbind(zASL,zTSL,zTAL,zbags0789,zbayarea,zgear))

```

9.1.4.2. Principal Components Analysis

```

library(FactoMineR)
PCA1<-zscores
PCAm <- as.matrix (PCA1)
PCA(PCAm)
##using prcomp for eigenvectors
pcatest <- prcomp(PCAm)
round <- PCA(PCAm) ### either cos2 or coord is the equation of the eigen vectors,
round$var
princomp(PCAm)
PRIN <- princomp(PCAm)
plot(PRIN)
pca1 <- prcomp(PCAm)
pca1$sdev^2/sum(pca1$sdev^2) ## percentages of variance explained by dimensions

```

9.1.5. Sensitivity Analysis

```

ASL <-as.numeric(eelgrass$active.suspension.lease..ha..2011)
eelgrass0789 <-rep(0,14)
ASL <- ASL/1.0237 ##### MIN ASL
predict.df <- data.frame(cbind(eelgrass0789,ASL,TSL,TAL,bayarea,gear,bags0789))
Var1 <- predict(mod1,newdata=predict.df,type="response")
print(predict.df)
print(Var1)
mean(Var1)
plot(sort(ASL),(Var1),type="l")

ASL <-as.numeric(eelgrass$active.suspension.lease..ha..2011)
eelgrass0789 <-rep(0,14)
ASL <- ASL*1.0237 ##### MAX ASL

```



```

predict.df <- data.frame(cbind(eelgrass0789,ASL,TSL,TAL,bayarea,gear,bags0789))
Var2 <- predict(mod1,newdata=predict.df,type="response")
print(predict.df)
print(Var2)
mean(Var2)
plot(sort(ASL),(Var2),type="l")
ASL <-as.numeric(eelgrass$active.suspension.lease..ha..2011)

eelgrass0789 <-rep(0,14)
TSL <- TSL/1.0640 ##### MIN TSL
predict.df <- data.frame(cbind(eelgrass0789,ASL,TSL,TAL,bayarea,gear,bags0789))
Var3 <- predict(mod1,newdata=predict.df,type="response")
print(predict.df)
print(Var3)
mean(Var3)
plot(sort(TSL),(Var3),type="l")

TSL <-as.numeric(eelgrass$total.suspended.lease.area)
eelgrass0789 <-rep(0,14)
TSL <- TSL*1.0640 ##### MAX TSL
predict.df <- data.frame(cbind(eelgrass0789,ASL,TSL,TAL,bayarea,gear,bags0789))
Var4 <- predict(mod1,newdata=predict.df,type="response")
print(predict.df)
print(Var4)
mean(Var4)
plot(sort(TSL),(Var4),type="l")
TSL <-as.numeric(eelgrass$total.suspended.lease.area)

TAL <-as.numeric(eelgrass$X2011.total.lease..ha.)
eelgrass0789 <-rep(0,14)
TAL <- TAL/1.0813 ##### MIN TAL
predict.df <- data.frame(cbind(eelgrass0789,ASL,TSL,TAL,bayarea,gear,bags0789))
Var5 <- predict(mod1,newdata=predict.df,type="response")
print(predict.df)
print(Var5)
mean(Var5)
plot(sort(TAL),(Var5),type="l")
TAL <-as.numeric(eelgrass$X2011.total.lease..ha.)

eelgrass0789 <-rep(0,14)
TAL <- TAL*1.0813 ##### MAX TAL
predict.df <- data.frame(cbind(eelgrass0789,ASL,TSL,TAL,bayarea,gear,bags0789))
Var6 <- predict(mod1,newdata=predict.df,type="response")
print(predict.df)
print(Var6)
mean(Var6)
plot(sort(TAL),(Var6),type="l")
TAL <-as.numeric(eelgrass$X2011.total.lease..ha.)

bags0789 <-as.numeric (eelgrass$X2007.2009.bags)
eelgrass0789 <-rep(0,14)
bags0789 <- bags0789/1.0846 ##### MIN BAGS
predict.df <- data.frame(cbind(eelgrass0789,ASL,TSL,TAL,bayarea,gear,bags0789))
Var7 <- predict(mod1,newdata=predict.df,type="response")
print(predict.df)
print(Var7)

```

```

mean(Var7)
plot(sort(bags0789),(Var7),type="l")

bags0789 <-as.numeric (eelgrass$X2007.2009.bags)
eelgrass0789 <-rep(0,14)
bags0789 <- bags0789*1.0846 ##### MAX BAGS
predict.df <- data.frame(cbind(eelgrass0789,ASL,TSL,TAL,bayarea,gear,bags0789))
Var8 <- predict(mod1,newdata=predict.df,type="response")
print(predict.df)
print(Var8)
mean(Var8)
plot(sort(bags0789),(Var8),type="l")
bags0789 <-as.numeric (eelgrass$X2007.2009.bags)

gear <-as.numeric(eelgrass$Gear.area..ha..2011)
eelgrass0789 <-rep(0,14)
gear <- gear/1.0003 ##### MIN GEAR
predict.df <- data.frame(cbind(eelgrass0789,ASL,TSL,TAL,bayarea,gear,bags0789))
Var9 <- predict(mod1,newdata=predict.df,type="response")
print(predict.df)
print(Var9)
mean(Var9)
plot(sort(gear),(Var9),type="l")
gear <-as.numeric(eelgrass$Gear.area..ha..2011)

eelgrass0789 <-rep(0,14)
gear <- gear*1.0003 ##### MAX GEAR
predict.df <- data.frame(cbind(eelgrass0789,ASL,TSL,TAL,bayarea,gear,bags0789))
Var10 <- predict(mod1,newdata=predict.df,type="response")
print(predict.df)
print(Var10)
mean(Var10)
plot(sort(gear),(Var10),type="l")

```

9.2. R-Studio Code for Estuary-Scale Statistics

9.2.1. Loading Data

```

eelgrass <- estuaries.raw.07.09
eelgrass0789 <-as.numeric(eelgrass$X2007.2009.EC.DFO.eelgrass..ha.)
ASL <-as.numeric(eelgrass$active.suspension.lease..ha..2011)
TSL <-as.numeric(eelgrass$total.suspended.lease.area)
TBL <- as.numeric (eelgrass$X2011.bottom.lease..ha.)
TAL <-as.numeric(eelgrass$X2011.total.lease..ha.)
estarea <-as.numeric(eelgrass$Bay.Area..ha.)
bags0789 <-as.numeric (eelgrass$X2007.2009.bags)
gear <-as.numeric(eelgrass$Gear.area..ha..2011)

```

9.2.2. Multiple Regression Analysis

```

newestdata1 <-data.frame(cbind(ASL,TSL,TAL,estarea,bags0789,gear)) ##est scale data frame
cor(newestdata1) ##checking correlation in data
mod1 <- glm(eelgrass0789 ~
ASL+TSL+TAL+estarea+gear+bags0789,data=newestdata1,family="poisson")
summary(mod1)
step1 <-step(mod1)
summary(step1) #####dropped gear

```

```

step2 <- step(step1)
summary(step2) ### no more drops
mod2 <- glm(eelgrass0789 ~ ASL+TSL+TAL+estarea+bags0789,data=newestdata1,family="poisson")
summary(mod2)
plot(mod2)

```

9.2.3. Principal Components Analysis

9.2.3.1. Z-Score Transformations

```

mean(eelgrass0789)
var(eelgrass0789)
zeelgrass <- (eelgrass0789-mean(eelgrass0789))/sqrt(var(eelgrass0789))
summary(zeelgrass)
plot(zeelgrass)

```

```

mean(ASL)
var(ASL)
zASL <- (ASL-mean(ASL))/sqrt(var(ASL))
summary(zASL)
plot(zASL)

```

```

mean(TSL)
var(TSL)
zTSL <- (TSL-mean(TSL))/sqrt(var(TSL))
summary(zTSL)
plot(zTSL)

```

```

mean(TAL)
var(TAL)
zTAL <- (TAL-mean(TAL))/sqrt(var(TAL))
summary(zTAL)
plot(zTAL)

```

```

mean(bags0789)
var(bags0789)
zbags0789 <- (bags0789-mean(bags0789))/sqrt(var(bags0789))
summary(zbags0789)
plot(zbags0789)

```

```

mean(estarea)
var(estarea)
zestarea <- (estarea-mean(estarea))/sqrt(var(estarea))
summary(zestarea)
plot(zestarea)

```

```

mean(gear)
var(gear)
zgear <- (gear-mean(gear))/sqrt(var(gear))
summary(zgear)
plot(zgear)

```

```

zscores <- data.frame(cbind(zASL,zTSL,zTAL,zbags0789,zestarea,zgear))

```

11.2.3.2. Principal Components Analysis

```

library(FactoMineR)
PCA1 <- zscores

```

```

PCAm <- as.matrix (PCA1)
PCA(PCAm)
##using prcomp for eigenvectors
pcatest <- prcomp(PCAm)
round <- PCA(PCAm) ### either cos2 or coord is the equation of the eigen vectors,
round$var
princomp(PCAm)
PRIN <- princomp(PCAm)
plot(PRIN)
pca1 <- prcomp(PCAm)
pca1$sdev^2/sum(pca1$sdev^2) ## percentages of variance explained by dimensions

```

11.2.4. Sensitivity Analysis

```

ASL <-as.numeric(eelgrass$active.suspension.lease..ha..2011)
eelgrass0789 <-rep(0,8)
ASL <- ASL/1.0259 ##### MIN ASL
predict.df <- data.frame(cbind(eelgrass0789,ASL,TSL,TAL,bayarea,bags0789))
Var1 <- predict(mod2,newdata=predict.df,type="response")
print(predict.df)
print(Var1)
mean(Var1)
plot(sort(ASL),(Var1),type="l")

```

```

ASL <-as.numeric(eelgrass$active.suspension.lease..ha..2011)
eelgrass0789 <-rep(0,8)
ASL <- ASL*1.0259 ##### MAX ASL
predict.df <- data.frame(cbind(eelgrass0789,ASL,TSL,TAL,bayarea,bags0789))
Var2 <- predict(mod2,newdata=predict.df,type="response")
print(predict.df)
print(Var2)
mean(Var2)
plot(sort(ASL),(Var2),type="l")
ASL <-as.numeric(eelgrass$active.suspension.lease..ha..2011)

```

```

eelgrass0789 <-rep(0,8)
TSL <- TSL/1.0599 ##### MIN TSL
predict.df <- data.frame(cbind(eelgrass0789,ASL,TSL,TAL,bayarea,bags0789))
Var3 <- predict(mod2,newdata=predict.df,type="response")
print(predict.df)
print(Var3)
mean(Var3)
plot(sort(TSL),(Var3),type="l")

```

```

TSL <-as.numeric(eelgrass$total.suspended.lease.area)
eelgrass0789 <-rep(0,8)
TSL <- TSL*1.0599 ##### MAX TSL
predict.df <- data.frame(cbind(eelgrass0789,ASL,TSL,TAL,bayarea,bags0789))
Var4 <- predict(mod2,newdata=predict.df,type="response")
print(predict.df)
print(Var4)
mean(Var4)
plot(sort(TSL),(Var4),type="l")
TSL <-as.numeric(eelgrass$total.suspended.lease.area)

```

```

TAL <-as.numeric(eelgrass$X2011.total.lease..ha.)
eelgrass0789 <-rep(0,8)

```

```
TAL <- TAL/1.0793 ##### MIN TAL
predict.df <- data.frame(cbind(eelgrass0789,ASL,TSL,TAL,bayarea,bags0789))
Var5 <- predict(mod2,newdata=predict.df,type="response")
print(predict.df)
print(Var5)
mean(Var5)
plot(sort(TAL),(Var5),type="l")
TAL <-as.numeric(eelgrass$X2011.total.lease..ha.)
```

```
eelgrass0789 <-rep(0,8)
TAL <- TAL*1.0793 ##### MAX TAL
predict.df <- data.frame(cbind(eelgrass0789,ASL,TSL,TAL,bayarea,bags0789))
Var6 <- predict(mod2,newdata=predict.df,type="response")
print(predict.df)
print(Var6)
mean(Var6)
plot(sort(TAL),(Var6),type="l")
TAL <-as.numeric(eelgrass$X2011.total.lease..ha.)
```

```
bags0789 <-as.numeric (eelgrass$X2007.2009.bags)
eelgrass0789 <-rep(0,8)
bags0789 <- bags0789/1.0192 ##### MIN BAGS
predict.df <- data.frame(cbind(eelgrass0789,ASL,TSL,TAL,bayarea,bags0789))
Var7 <- predict(mod2,newdata=predict.df,type="response")
print(predict.df)
print(Var7)
mean(Var7)
plot(sort(bags0789),(Var7),type="l")
```

```
bags0789 <-as.numeric (eelgrass$X2007.2009.bags)
eelgrass0789 <-rep(0,8)
bags0789 <- bags0789*1.0192 ##### MAX BAGS
predict.df <- data.frame(cbind(eelgrass0789,ASL,TSL,TAL,bayarea,bags0789))
Var8 <- predict(mod2,newdata=predict.df,type="response")
print(predict.df)
print(Var8)
mean(Var8)
plot(sort(bags0789),(Var8),type="l")
bags0789 <-as.numeric (eelgrass$X2007.2009.bags)
```

9.3. R-Studio Code for Watershed-Scale Statistics

9.3.1. Loading Data

```
eelgrass <- estuaries.raw.07.09
eelgrass0789 <- as.numeric (eelgrass$X2007.2009.EC.DFO.eelgrass..ha.)
AGR<-as.numeric(eelgrass$AGR)
IND<-as.numeric(eelgrass$IND)
INF<-as.numeric(eelgrass$INF)
SET<-as.numeric(eelgrass$SET)
Forest<-as.numeric(eelgrass$Forested.Area..ha.)
nonforest <-as.numeric(eelgrass$non.forested.area..ha.)
watershed <- as.numeric (eelgrass$Watershed.Area..ha.)
```

9.3.2. Principal Components Analysis

11.3.2.1. Z-Score Transformations

```
mean(eelgrass0789)
var(eelgrass0789)
zeelgrass0789 <- (eelgrass0789-mean(eelgrass0789))/sqrt(var(eelgrass0789))
summary(zeelgrass0789)
plot(zeelgrass0789)
```

```
mean(AGR)
var(AGR)
zAGR <- (AGR-mean(AGR))/sqrt(var(AGR))
summary(zAGR)
plot(zAGR)
```

```
mean (IND)
var(IND)
zIND <- (IND-mean(IND))/sqrt(var(IND))
summary(zIND)
plot(zIND)
```

```
mean (INF)
var(INF)
zINF <- (INF-mean(INF))/sqrt(var(INF))
summary(zINF)
plot(zINF)
```

```
mean (SET)
var(SET)
zSET <- (SET-mean(SET))/sqrt(var(SET))
summary(zSET)
plot(zSET)
```

```
mean (Forest)
var(Forest)
zForest <- (Forest-mean(Forest))/sqrt(var(Forest))
summary(zForest)
plot(zForest)
```

```
mean (watershed)
var(watershed)
zwatershed <- (watershed-mean(watershed))/sqrt(var(watershed))
summary(zwatershed)
plot(zwatershed)
```

```
zscores <-data.frame (cbind(zAGR,zIND,zINF,zSET,zForest,zwatershed))
```

11.3.2.2. Principal Components Analysis

```
library(FactoMineR)
PCA1<-zscores
PCAm <- as.matrix (PCA1)
PCA(PCAm)
##using prcomp for eigenvectors
pctest <- prcomp(PCAm)
```

```
round <- PCA(PCAm)
round$var
princomp(PCAm)
PRIN <- princomp(PCAm)
plot(PRIN)
pca1 <- prcomp(PCAm)
pca1$sdev^2/sum(pca1$sdev^2)
```