SOCIOECONOMIC RISK FROM OCEAN ACIDIFICATION AND CLIMATE CHANGE IMPACTS ON ATLANTIC CANADIAN FISHERIES

by

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DEDICATION

For Britt who has stuck with me from one coast to the other, and who kept me going through the good days and the long days. I can't wait for our next adventure!
For my mom who has always encouraged me to do my best, and to be proud of what I have accomplished.
For my grandfathers who I wish were here to celebrate with me.
And for all my friends and family who have supported me, not only through the past couple years, but throughout my life.

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ABSTRACT

Ocean acidification (OA) is an emerging consequence of anthropogenic carbon dioxide emissions. The full extent of the biological impacts are currently not well understood. However, it is expected that invertebrate species that rely on the mineral calcium carbonate will be among the first and most severely affected. Despite the limited understanding of impacts there is a need to identify potential pathways for human societies to be affected by OA. Research on these social implications is a small but developing field of literature. This thesis contributes to this field by using a risk assessment framework, informed by a biophysical model of future species distributions, to investigate Atlantic Canadian risk from changes in shellfish fisheries. New Brunswick and Nova Scotia are expected to see declines in resource accessibility. While Newfoundland and Labrador and PEI are more socially vulnerable to losses in fisheries, they are expected to experience relatively minor changes in access.

LIST OF ABBREVIATIONS AND SYMBOLS USED

Ω	Saturation state of calcium carbonate
Ca^{2+}	Calcium ion
$CaCO_3$	Calcium carbonate
CO_2	Carbon Dioxide
CO_3^{2-}	Carbonate ion
DBEM	Dynamic Bioclimate Envelope Model
DFO	Fisheries and Oceans Canada (formerly Department of Fisheries and Oceans)
EEZ	Exclusive Economic Zone
FAO	Food and Agriculture Organization
GDP	Gross Domestic Product
GFDL	Geophysical Fluid Dynamics Laboratory
H^+	Hydrogen ion
H_2CO_3	Carbonic acid
HCO_3^-	Bicarbonate ion
IPCC	Intergovernmental Panel on Climate Change
IPSL	Institute Pierre Simon Laplace Climate Modelling Centre
K_{sp}	Solubility constant
LFA	Lobster Fishing Area
LIFO	Last In First Out
MPI	Max Planck Institute for Meteorology
NAFO	Northwest Atlantic Fisheries Organization
NB	New Brunswick
NL	Newfoundland and Labrador
NOAA	National Oceanic and Atmospheric Administration
NS	Nova Scotia
OA	Ocean Acidification
PEI	Prince Edward Island
PPM	Parts Per Million
Que	Quebec
RCP	Representative Concentration Pathway
SVR	Social Vulnerability and Risk
UBC	University of British Columbia

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Chapter 1 Introduction

1.1 Framing This Thesis

The goal of this Master of Environmental Studies thesis is to make a preliminary assessment of the impacts of ocean acidification (OA) and climate change-related phenomena on commercially valuable fisheries in Atlantic Canada and the potential resulting impact on dependent communities. The work is interdisciplinary, merging and analysing data from both natural and social science domains. The conclusions are framed in a social context and presented as a risk assessment, wherein the risk posed to provinces in Atlantic Canada are scored relative to each other based on three components of risk (exposure, sensitivity and adaptive capacity). However, the risk assessment is founded on a biophysical model projection of expected species responses to OA and climate change through the 21st century.

This introductory chapter provides essential information regarding the diverse concepts (i.e., OA and climate change, fisheries, biophysical modelling and risk assessment theory) that have been brought together to assess the risk posed by ocean and fisheries changes to the Atlantic Canadian region. Chapter 2 will present a review of methods used in other localities to assess the social and economic impacts that may result specifically from OA. Chapter 3 includes the methods and main findings of this thesis. The final chapter will expand on the implications (and shortcomings) of the results and discuss future directions for research related to OA and climate change as they apply to fisheries. Chapters 2 and 3 are written as independent chapters for potential publication. There is therefore some overlap of content in their respective introductions.

1.2 Climate Change and Ocean Acidification

"Human influence on the climate system is clear, and recent anthropogenic emissions of greenhouse gases are the highest in history. Recent climate changes have had widespread impacts on human and natural systems"
(IPCC, 2014, p. 2)

Since the pre-industrial era, human emissions of carbon dioxide (CO_2) have raised atmospheric concentrations from around 280 parts per million (ppm) to over 400 ppm (IPCC, 2014; NOAA, 2017). Atmospheric CO_2 acts as a greenhouse gas and is a key driver of global climate change. To limit the global average temperature increase to less than 2°C (a global target chosen to encourage international cooperation in reducing CO_2 emissions in hopes of avoiding the most drastic impacts associated with climate change), atmospheric CO_2 should not exceed 430 ppm (IPCC, 2014). Various factors affect emissions, including population size, economic activity and lifestyle (IPCC, 2014). Scenarios describing the interactions of these factors and future CO_2 emission trajectories have been condensed into four Representative Concentration Pathways (RCPs); each RCP scenario is the product of many separate emissions projections (Figure 1). The number associated with each RCP is indicative of the radiative forcing expected by 2100 (in W/m^2) (van Vuuren et al., 2011). These scenarios can be used in climate models to generate predictions of future climate conditions out to the year 2100 (Figure 1). RCP 2.6 is the lowest emission scenario, where emission rates decrease dramatically and crossing the 2°C temperature threshold for change is unlikely. At the other extreme, RCP 8.5 is the highest emission scenario and represents uncontrolled growth of CO_2 emissions through the rest of the century (IPCC, 2014; van Vuuren et al., 2011).

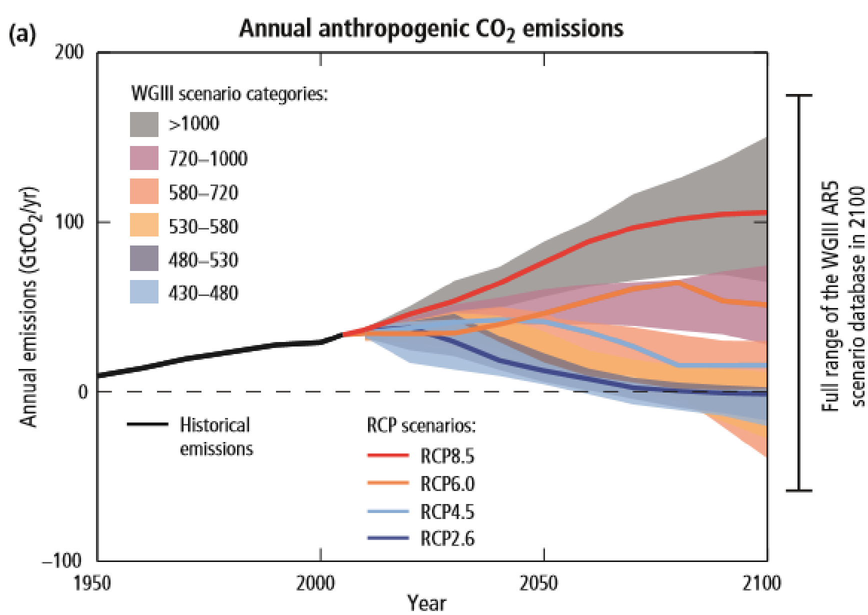

Figure 1. RCP emission scenarios from (IPCC, 2014, p. 9).The solid lines are the annual CO_2 emissions for the indicated RCP scenario, while the shaded area surrounding each of these represents the range of scenarios that contributed to the RCPs (WGIII refers to Working group III of the IPCC Fifth Assessment Report).

Many of the expected environmental and ecological impacts of climate change will directly impact human health and wellbeing. Climate change is often thought of in terms of temperature change (i.e., global warming), and warming is also the main driver for many other effects of climate change (e.g., sea-level rise due to melting ice; changing weather patterns due to changing temperature gradients; species migrations to follow biologically suitable temperature ranges (Hoegh-Guldberg et al., 2014; IPCC, 2014)). However, another effect of increasing atmospheric CO_2 is (almost) completely independent of temperature: OA is the result of changing seawater chemistry. Increasing concentrations of atmospheric CO_2 drive aquatic uptake of CO_2, and this causes the pH of water to decrease (i.e., become more acidic[1]). While the physical processes causing temperature change and OA are independent of each other, the two factors happen concurrently and have the potential to have interactive and even compounding impacts on marine species and ecosystems (e.g., Byrne & Przeslawski, 2013; Kroeker, Kordas, & Harley, 2017; Portner, 2012).

1.2.1 Ocean chemistry

As anthropogenic CO_2 is emitted into the atmosphere, a portion of it is absorbed by the ocean. Currently 25-30% of anthropogenic CO_2 emitted each year is taken up by seawater (Feely et al., 2004; Khatiwala et al., 2013; Sabine et al., 2004). This has had a mitigating effect on climate change by reducing the total CO_2 acting as a greenhouse gas in the atmosphere, but has resulted in a change in seawater chemistry (Orr et al., 2005). The ocean surface absorbs CO_2 from the atmosphere through diffusion driven by the greater partial pressure of CO_2 in the atmosphere than in the ocean. In the absence of human perturbation, the surface ocean and the atmosphere would reach equilibrium with respect to CO_2 on a time scale of months. However, equilibrium will not be reached as long as concentrations in the atmosphere continue to rise (Sabine et al., 2004). Over much longer timescales, because of deep ocean circulation, the whole ocean is expected to have the capacity to absorb up to 90% of anthropogenic CO_2, though this would have drastic implications for the chemistry of the ocean (e.g., Sabine et al., 2004). The preindustrial

[1] Note 'more acidic' and 'acidification' are relative terms and do not inherently mean a substance is an acid. This is similar to how using 'warmer' to describe a change in temperature does not necessarily mean the temperature is 'warm'. A pH of 7.9 is 'more acidic' than 8.1, but it is still alkaline.

global average ocean pH was around 8.2, which has since declined to 8.1 (e.g., Royal Society, 2005; Washington State, 2012), and is projected to fall by as much as another 0.3 to 0.4 units by the end of the century (Orr et al., 2005) assuming unchecked CO_2 emissions.

In seawater, CO_2 molecules react with water molecules (H_2O) to form carbonic acid (H_2CO_3), resulting in a change in pH. Seawater naturally buffers this pH change through a series of chemical reactions referred to as the carbonate system. The carbonate system is comprised of carbonic acid (H_2CO_3), bicarbonate ion (HCO_3^-) and carbonate ion (CO_3^{2-}) (Equation 1; Figure 2). These molecules reach equilibrium in seawater, with HCO_3^- being the most common form for pH values near that of recent historical seawater; however, increasing the concentration of CO_2 shifts the equilibrium toward H_2CO_3 (Figure 2) (Doney, Fabry, Feely, & Kleypas, 2009; Orr et al., 2005). While this shift moderates the change in pH, it results in a significant decrease in the concentration of CO_3^{2-}, which has been demonstrated to impact marine species that extract calcium carbonate ($CaCO_3$) from seawater to form shells and exoskeletons. This includes, but is not limited to corals, molluscs and crustaceans.

Equation 1.

$$CO_2(atm) \rightleftharpoons CO_2(aq) + H_2O \rightleftharpoons H_2CO_3 \rightleftharpoons H^+ + HCO_3^- \rightleftharpoons 2H^+ + CO_3^{2-}$$

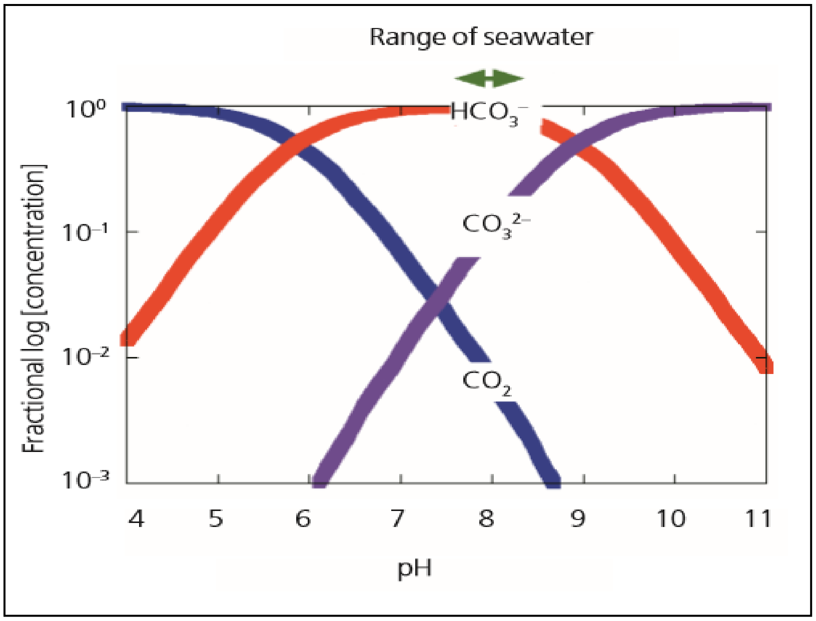

Figure 2. Relative concentrations of the carbonate system molecules. As total carbon in the system increases, the pH is also pushed towards the left side of the figure and ionic concentrations change according to the plot. The green double arrow at the top of the figure indicates normal range of pH in seawater. The relationship between the system components is explained by Equation 1. Figure from Royal Society (2005, pg.6).

1.2.1.1 Coastal Systems

Changes in pH and CO_2 concentrations are further complicated in coastal systems where both natural factors and human activity interact with seawater chemistry (Doney, 2010; Duarte et al., 2009; Washington State, 2012). Factors such as freshwater inputs, upwelling and eutrophication can all locally draw down pH and exacerbate the background effects of atmospheric CO_2-driven OA (Doney, 2010; Duarte et al., 2009; Washington State, 2012). The effect from coastal factors is so significant that Duarte et al. (2009) propose that it should be classified and discussed as a separate (albeit related) phenomenon from open-ocean OA. Understanding how all these factors chemically interact to affect pH is complicated, and current models projecting future OA scenarios generally do not account for these local factors.

1.2.1.2 Calcium Carbonate

$CaCO_3$ is a mineral made from calcium ions (Ca^{2+}) and CO_3^{2-}, both of which are readily available in seawater under normal conditions. The stability of $CaCO_3$ minerals is described by the saturation state of the solution (Ω). The Ω is defined by the concentrations of Ca^{2+} and CO_3^{2-}, in relation to the solubility constant (k_{sp}) for the specific mineral under consideration (Equation 2) (Branch, DeJoseph, Ray, & Wagner, 2013; Doney et al., 2009; Kleypas et al., 1999). The main driver of changes in Ω in the ocean is changes in the concentration of CO_3^{2-} in solution. This is because in seawater, the concentrations of Ca^{2+} are fairly and relatively over-abundant. Temperature, pressure, and salinity also play a role in Ω, as they affect the k_{sp}; however, the impact on the overall Ω is small compared to the effect of changes in concentrations of CO_3^{2-} (Branch et al., 2013; Doney et al., 2009; Feely et al., 2004; Kleypas et al., 1999; Mucci, 1983).

Equation 2

$$\Omega_{sp} = \frac{[\mathrm{Ca}^{2+}] \cdot [\mathrm{CO}_3^{2-}]}{k_{sp}}$$

$CaCO_3$ comes in several crystal forms. The two most common and biologically relevant are calcite and aragonite. Aragonite is the most common form of calcium carbonate found in the shells of molluscs, while calcite is the primary form of calcium carbonate found in crustacean exoskeletons (though some molluscs also rely on calcite) (Branch et al., 2013). These two crystal forms have different properties, including their k_{sp} and hence their specific Ω – aragonite is less stable than calcite, and therefore dissolves much more readily (Branch et al., 2013; Doney et al., 2009; Orr et al., 2005). The global average Ω for surface waters for aragonite is 2-4, while for calcite the average Ω is 4-6, (Branch et al., 2013; Kleypas et al., 1999). As more CO_2 dissolves into the ocean average ocean Ω for both minerals will decrease. When Ω is less than 1, unprotected calcium carbonate structures will dissolve; when Ω is greater than 1, calcium carbonate crystals can precipitate out of solution (Branch et al., 2013; Kleypas et al., 1999; Morel & Hering, 1993; Orr et al., 2005).

1.2.1.3 Biologic Impacts

Even while Ω values are greater than 1, if they fall below the levels normally experienced by organisms, calcification rates can decrease and/or energy demands for calcification will increase. This could result in reduced growth rates, higher susceptibility to predation and less metabolic energy for other activities (e.g., Branch et al., 2013; Doney et al., 2009; Kleypas et al., 1999; Portner, 2012). There is evidence that adult molluscs and crustaceans may experience decreased growth, calcification, and survival in response to

OA; with molluscan species tending to show more pronounced responses (Kroeker et al., 2013; Kroeker, Kordas, Crim, & Singh, 2010). Importantly, larval stages of both taxa appear to be much more severely affected by OA. Initial formation of $CaCO_3$ shells and subsequent metamorphosis between life-stages incur high energy demands. The extra costs associated with a decreased Ω has resulted in higher larval mortalities in laboratory studies (e.g., Frieder, Applebaum, Pan, Hedgecock, & Manahan, 2016; Kroeker et al., 2013, 2010; Portner, 2012; Spalding, Finnegan, & Fischer, 2017).

However, species-specific responses to OA in individual studies have been highly variable between related species, and even within populations of the same species. It is likely that a variety of other environmental factors (e.g., food availability and temperature), as well as specific genetic traits of a population affect sensitivity to OA (Branch et al., 2013; Kroeker et al., 2013, 2010).

Finfish, which do not rely on $CaCO_3$ hard parts, were originally expected to be generally immune to direct effects of OA. However, some studies have indicated that decreased pH may affect them in ways that include navigation, predator avoidance, fertilization and larval development (e.g., Branch et al., 2013; Dixson, Munday, & Jones, 2010; Kroeker et al., 2013; Murray, Fuiman, & Baumann, 2016). This suggests that OA may have much wider ranging impacts than anticipated. Nonetheless, given the more direct relationship between calcifying species and OA, these species remain the focus for the majority of research conducted to date.

1.2.1.4 Interactions with Other Climate Stressors

OA will co-occur with a host of other climate change factors that will affect biology. Most notable and ubiquitous among these is temperature change; however, factors such as oxygen depletion and changing salinities will also affect behaviours and physiologies (Hoegh-Guldberg et al., 2014). The physiological impacts of these combined environmental factors are unlikely to be straightforward and may interact in a variety of ways (e.g., additive or synergistic interactions) (Kroeker et al., 2017). Predictably, given the lack of complete understanding of the effects of OA, there is currently an even larger gap in understanding of how organisms are likely to respond to combined stressors. Kroeker et al. (2017) suggest that a more thorough understanding of the physiological impacts of OA on its own will, however, allow theoretical approaches to begin to predict how multiple stressors may interact.

Besides the physical and chemical interactions and synergies that affect individual organisms, there are also likely to be interactions between species which will ripple through food webs and ecosystems (Gaylord et al., 2015; Kroeker et al., 2017). Predictions regarding ecological impacts require a thorough understanding of both the ecosystem being considered as well as the potential effect of the stressor (i.e., OA) on all relevant species. Work with the Atlantis model (Fulton, Parslow, Smith, & Johnson, 2004) makes a foray into this field by accounting for ecological relationships; however, detailed predictions are still constrained by the currently limited OA response data (Kaplan, Levin, Burden, & Fulton, 2010; K. N. Marshall et al., 2017)

1.2.1.5 Socioeconomic Analyses

Following the relatively recent recognition of OA as a global phenomenon with the potential for significant ecosystem impacts, the amount of research dedicated to the topic has expanded rapidly. Two meta-analyses of biological responses to OA, the first conducted in 2010 and the second in 2013, saw an increase from 73 to 228 studies that fit their criteria for analysis, representing nearly a threefold increase over a three year period (Kroeker et al., 2013, 2010). With the growing scientific interest in the subject, there has also been an increasing interest in how OA might affect human communities that use or benefit from marine resources. The majority of these studies consider how OA is likely to affect future fisheries landings and how these changes could affect human communities (e.g., Cooley, Lucey, Kite-Powell, & Doney, 2012; Mathis et al., 2015; Narita, Rehdanz, & Tol, 2012); however, a subset also consider less direct pathways for impacts on human communities such as through tourism-related losses and changes to other ecosystem services (e.g., L. M. Brander, Rehdanz, Tol, & Van Beukering, 2012). To date the majority of the socioeconomic assessments of OA have addressed potential future OA impacts in the absence of other climate factors. This may stem from the emerging nature of the subject and a lack of a clear understanding on how OA driven changes will affect broader social and ecological systems. By focusing solely on the OA effect, researchers can begin to understand how OA will interplay with other global scale changes. It is, however, important to acknowledge that in reality OA will not occur in isolation. While the manner in which OA and other climate stressors will interact is still largely unknown and difficult to quantify, it should nonetheless be a key feature of discussions and investigations into OA related studies (Kroeker et al., 2017).

1.3 Fisheries

Fisheries are an important source of protein for a significant portion of the global population, with over 3 billion people getting more than 15% of their dietary protein from aquatic sources. There are further nutritional benefits stemming from some critical micro-nutrients (e.g., omega-3 fatty acids) that are more readily available in seafood than terrestrially sourced foods (FAO, 2016; Golden et al., 2016). The availability of these nutrients through seafood is often much more relevant in developing nations where alternative sources are not always easily accessible; however, these nutrients can be regionally important in some developed nations as well (FAO, 2016; Golden et al., 2016). Their nutritional importance makes fisheries and aquaculture an integral part of global food security. As with many other food systems, seafood production is likely to be impacted by climate change (Blanchard et al., 2017; FAO, 2016; Golden et al., 2016); therefore, understanding how these changes are likely to manifest can help to mitigate the impacts and allow for pro-active action.

1.3.1 Atlantic Canadian Fisheries

For Canada (as well as most other developed nations), where food security is not typically a national concern and many sources of protein are readily available, marine resources are primarily relevant in terms of economic value and employment opportunities for coastal communities (FAO, 2016; Smith et al., 2010). Although, there are instances, especially in more remote regions which have limited food purchasing options, where marine resources are essential food sources (Berkes, 1990; Divovich, Belhabib, Zeller, & Pauly, 2015).

Roughly half of all Canadian marine fishery landings are shellfish (in 2015, shellfish represented 449,000T out of a total of 818,000T; Figure 3). Canada has major fishery operations on its Pacific and Atlantic coasts with a combined annual landed value over $3 billion (Figure 3). However, the vast majority of Canadian landings are currently (and were historically) taken from the Atlantic coast (Figure 3). This includes ~95% of Canadian shellfish, although the Atlantic coast also out-produces the Pacific coast for other species groups (i.e., pelagic finfish and groundfish; Figure 3) (DFO, 2017c).

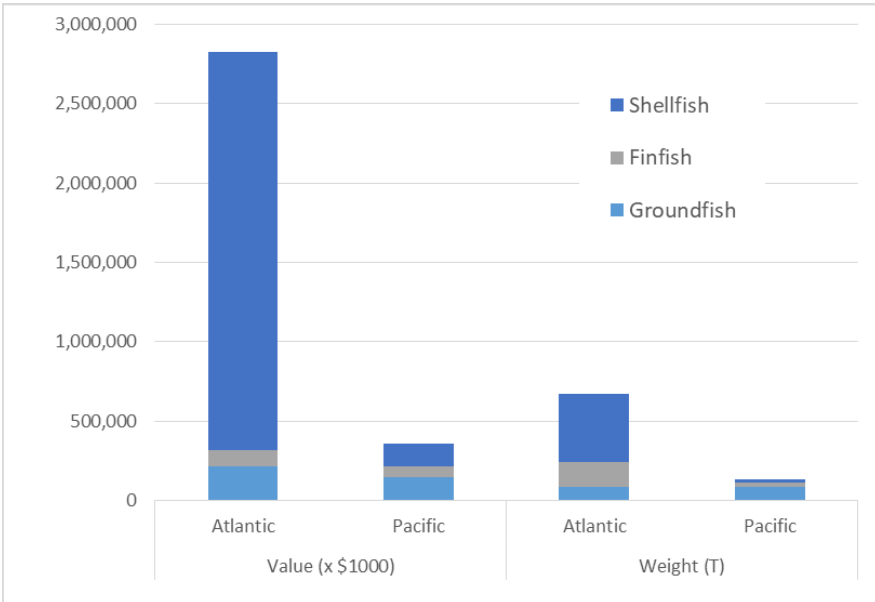

Figure 3. Canadian marine fisheries by coast. Left-hand section is the dollar value by species type for Atlantic and Pacific coasts; values are in thousands of dollars. Right-hand section is landing weights by species type; values are in tonnes. Data is from 2015 harvests (DFO, 2017c). Note DFO defines 'finfish' as pelagic vertebrate fishes, and 'groundfish' as demersal vertebrate fishes. 'Shellfish' encompasses all marine invertebrate species.

Following the collapse the Atlantic cod, and other groundfish, fisheries in the early 1990s, and subsequent change in ecosystem structure, harvest of shellfish species expanded significantly (Figure 4) (Dawe, Koen-Alonso, Chabot, Stansbury, & Mullowney, 2012; Divovich et al., 2015). The shift to invertebrate species has led to American lobster (*Homarus americanus*) becoming Canada's most valuable fishery (both in terms of landings value, and export value) (DFO, 2017c, 2017f). Furthermore, four of the five most valuable seafood exports are shellfish (American lobster, snow crab (*Chionoecetes opilio*), northern shrimp (*Pandalus* borealis) and sea scallop (*Placopecten magellanicus*)), the vast majority of which are sourced from Atlantic Canada. The only finfish in the top five exports is (farmed) Atlantic salmon (*Salmo salar*) (DFO, 2017c, 2017f).

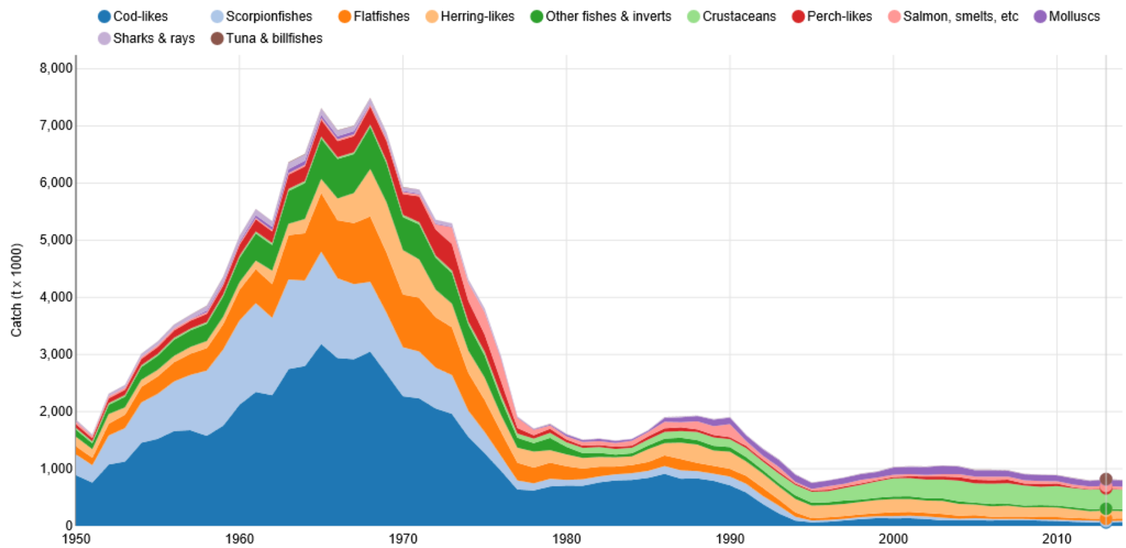

Figure 4. Reconstructed Canadian Atlantic historic catches, demonstrating the decline of groundfish landings ('cod-likes', 'scorpionfishes' and 'flatfishes'), and the increase of crustaceans (light green) and molluscs (dark purple). Figure and data from SeaAroundUs Project (Divovich et al., 2015; Zeller & Pauly, 2015).

With the expected strong susceptibility of invertebrate species to OA (Kroeker et al., 2013), the considerable importance of shellfish in Atlantic Canada presents a situation where the fishing industry as it currently exists is potentially at risk. This has not gone unnoticed by policy makers as well as the public at large – multiple news stories have covered the subject in recent years (e.g., DFO, 2014; South, 2016; Wangersky, 2015).

Furthermore, due to faster dissolution of CO_2 in colder waters, OA is expected to occur faster at higher latitudes (Fabry, McClintock, Mathis, & Grebmeier, 2009). This suggests that that Canadian waters may experience faster changes in pH than other oceanic regions. Conversely, changing temperature regimes are expected to result in poleward migration of species (Cheung et al., 2010). Which means Canada's temperate latitude may allow for increased species abundances. These two opposite trends have the potential to interact in Canadian Atlantic waters, with OA pressuring species away from the North, while temperature change drives species away from the South. The future of fisheries in the region will depend on the strength of the signals, the susceptibility of species to each stressor, and perhaps most importantly, how these stressors will interact.

1.3.1.1 Aquaculture and Capture Fisheries

For most invertebrate species in Atlantic Canada, the vast majority of production comes from wild harvest (DFO, 2017a, 2017c, 2017f). This is especially true for crustacean species where there is little if any cultured augmentation of wild populations, and no complete life-history culture. In partial contrast, approximately 25% of mollusc production in Atlantic Canada comes from aquaculture (DFO, 2017a, 2017f). Significant aquaculture production is limited to eastern blue mussels (*Mytilus edulis*) and American oysters (*Crassostrea virginica*). Overall, these species represent a relatively small fraction of the total harvest, but they are regionally relevant – especially mussel culture in the province of Prince Edward Island (PEI). However, aquaculture is a rapidly expanding industry both internationally and within Canada (DFO, 2013; Diana, 2009; FAO, 2016). Currently, much of the Canadian expansion is focused on finfish aquaculture. Although, there is controversy and lacklustre public support surrounding this type of aquaculture in much of Atlantic Canada (Doelle & Lahey, 2014), it is likely that shellfish culture will also see increased production.

Aquaculture production on the Pacific Coast of North America (i.e., Washington and Oregon states, as well as British Columbia), have already experienced significant losses due to OA-related events[2]. Recognition of the problem has allowed for adaptation of production, especially in hatcheries where the high-mortality events were first encountered. Water chemistry is now more closely monitored and controlled while the most susceptible life-stages develop (Clements & Chopin, 2016; Washington State, 2012). Narita, Rehdnaz & Tol (2012) suggest that these mitigation activities can allow for aquaculture production to be more resilient to OA compared to wild harvest. Nonetheless, they also note that in some regions hatcheries are not economically feasible. In these situations aquaculture production is reliant on wild populations, and is therefore likely no more secure than wild harvest ((Pillay & Kutty, 2005) in (Narita et al., 2012)). Much of the shellfish aquaculture production on Canada's West Coast includes hatchery rearing through larval stages. However, Atlantic Canadian shellfish aquaculture currently has limited hatchery capacity (Isabelle Tremblay, personal communication, November 3,

[2] These were mainly driven by upwelling – one of the local factors which affects pH and can compound OA.

2017) and is largely reliant on wild sources for recruitment, further exacerbating the region's potential susceptibility to OA.

1.4 Dynamic Bioclimate Envelope Model

The Dynamic Bioclimate Envelope Model (DBEM) was originally built by William Cheung and colleagues at the Changing Ocean Research Unit at the University of British Columbia (UBC) to assess the impact of climate change on marine species' global distributions (Cheung, Lam, & Pauly, 2008). To date the model results have been used to assess total changes in fisheries for large ocean regions (Cheung et al., 2010; Cheung, Watson, & Pauly, 2013; Cheung, Close, Lam, Watson, & Pauly, 2008). Initially, the model did not account for OA and focused mainly on other factors affecting life-history and survival that are driven by climate change, such as changes in temperature and dissolved oxygen. Cheung, Dunne, Sarmiento, and Pauly (2011) modified the model inputs to include OA effects, and Lam, Cheung and Sumaila (2014) used the DBEM with OA to assess the impacts on fisheries in the Arctic Ocean. Tai, Harley and Cheung (In prep.) are continuing development of the model, and improving the implementation of OA effects in the DBEM. The following section is a brief description of the model inputs and processes. For a complete description of the model see (Cheung et al., 2016, 2011; Cheung, Lam & Pauly, 2008).

The DBEM uses current species distributions to characterise preferred environmental conditions, and combines this with projections of future ocean conditions to predict future species distributions. This is accomplished by linking multiple models, starting with a species distribution model of current distributions with respect to depth, temperature, salinity, latitudinal range, and habitat type preferences (Close et al., 2006) to infer preferred environmental conditions on a half-degree latitude by half-degree longitude grid. Population dynamics are driven by a logistic growth model, and physiology is affected by changes in temperature, oxygen concentration and pH through a von Bertalanffy growth model ((von Bertalanffy, 1951) *in* (Cheung et al., 2011; Cheung, Lam & Pauly, 2008)). Movement between cells is directed by habitat suitability, which changes through time based on global climate models, which are in turn informed by the RCP scenarios for future CO_2 emissions. The DBEM then uses the predicted changes in species distributions along with modelled primary production to calculate future catch potential (an estimate of maximum sustainable yield) for each grid cell.

In the DBEM outputs used for this thesis, the two most extreme RCP scenarios were used (RCP 2.6 – highly reduced emissions; RCP 8.5 – continued expansion of emissions), to drive three separate global climate models (NOAA's Geophysical Fluid Dynamics Laboratory (GFDL) model; Max Planck Institute for Meteorology (MPI) model; and Institute Pierre Simon Laplace Climate Modelling Centre (IPSL) model). The OA effects were derived from the mean impact levels from Kroeker et al. (2013) and were implemented as effects on growth and survival for mollusc and crustacean species. The level of impacts were tied to changes in hydrogen ion concentrations (i.e., pH) (Lam et al., 2014; Tai et al., In prep.).

1.5 Risk and Vulnerability Assessments

About a third of the socioeconomic assessments of OA impacts conducted to date have approached the topic following risk or vulnerability assessment methodologies. These methods attempt to identify human communities which are susceptible to changes in a resource based on a range of societal factors. This approach was followed in this thesis, as it presents a more holistic understanding of the challenges posed to the Atlantic region, compared to alternative methods which address purely economic factors. Furthermore, the risk assessment results were anticipated to offer conclusions which would be more relevant to resource users and managers. Chapter 2 of this thesis presents a more detailed review of the various methods used to assess social impacts driven by OA and climate change; the following section is an introduction to the broader risk and vulnerability assessment theory and methodology to support that chapter.

Cardona et al. (2012, p. 91) present the International Organization for Standardization definition of a risk assessment as "a process to comprehend the nature of risk and to determine the level of risk." Risk assessments have often been applied to the risks posed by extreme events (e.g., tsunamis and earthquakes); however, there has been recent acknowledgment of the potential for applying the methods to more gradual, long-term pressures such as those presented by climate change (e.g., sea level rise) (e.g., N. A. Marshall et al., 2009). Risk assessments provide a framework for investigating the risks posed to communities, regions or nations from an array of hazards. The primary concern of risk assessments is usually dependent human communities; although, risk for human communities has been suggested to be inextricably linked to risks posed to ecological systems since changes therein can affect stability and access to resources (N. A. Marshall et al., 2009).

Due to the unique risks presented by combinations of specific regions and the various hazards that impact them, there is no single framework that fits all scenarios and geographies. Nonetheless, there is abundant literature regarding general outlines and considerations that should be followed when constructing a risk assessment (e.g., Cardona et al., 2012; Hobday et al., 2011; N. A. Marshall et al., 2009; Turner et al., 2003). All risk assessments contain the same three to five components; however, between assessments there are subtle differences in the definitions of these terms, as well as less subtle differences in their organization and linkages.

1.5.1 Terminology

An important issue that arises within risk assessment literature is the large degree of ambiguity in the definition of key terms. Each specific field that uses risk assessments (or even each specific set of authors within sub-fields), while using similar key terms, tends to use varying definitions of these terms. This is partly related to the different theoretical backgrounds of the various authors (ecology, social science, etc.) who bring their respective tendencies, habits and focuses (Gallopín, 2006). These discrepancies in definitions are recognized as an obstacle to the field, which can make it difficult for researchers from other fields to access or comprehend assessments. Additionally it leads to challenges when comparing results between studies. Nonetheless, there remains a lack of consensus regarding definitions in the field (Cardona et al., 2012; Gallopín, 2006).

Setting aside the definitional challenges (while also attempting not to compound them), key components in any risk assessment are: *exposure, vulnerability, sensitivity, adaptive capacity* and the *hazard* itself (Table 1). Many frameworks include *sensitivity* and *adaptive capacity* as sub-components of *vulnerability* (e.g., Ekstrom et al., 2015; Mathis et al., 2015; Turner et al., 2003); however, this is not universal (e.g., Johnson, Bell, & Gupta, 2016; N. A. Marshall et al., 2009; Turner et al., 2003). Furthermore, some frameworks use variations on the terminology – for example, using resilience in place of *adaptive capacity* (although resilience also has its own specific applications under some definitions and frameworks – Table 1), or susceptibility in place of *sensitivity* (Cardona et al., 2012).

Table 1. Definitions of key risk assessment terms from sources representing an array of specific definitions.

Term ↓ Source →	Cardona *et al.,* 2012	Marshall *et al.,* 2009	Turner *et al*., 2003
HAZARD	Possible future occurrence of events that may have adverse effects.	*[Not defined.]*	Threats to a system, resulting from perturbations and/or stress/stressors and the resultant consequences.
EXPOSURE	Elements in an area where a *hazard* may occur.	Extent to which a region, resource or community experiences changes (IPCC, 2007). Magnitude, frequency, duration and/or extent of an event (*hazard*).	*[Not defined.]*
VULNERABILITY	Propensity for exposed elements to be adversely affected by a *hazard*.	Degree to which a system is susceptible to, or unable to cope with adverse effects (IPCC, 2007). A function of character, magnitude and rate of change a system is *exposed* to & the system's sensitivity and adaptive capacity.	Degree to which a system will be impacted due to *exposure* to a *hazard*.
SENSITIVITY	Seen as a component of *vulnerability*. Physical predisposition of elements to be affected by a *hazard*.	Degree to which a system is affected by or responsive to climate change. Determined by level of dependence on specific good or service that will be affected.	Determined by human and environmental conditions of a system. (Formerly referred to as 'dose-response.')
ADAPTIVE CAPACITY	Seen as a component of *vulnerability*. Ability to access and mobilize resources in response to a *hazard* (proactively = 'adaptive capacity; reactively = 'coping capacity').	Ability to respond to challenges.	('*coping* capacity') Ability to respond to a *hazard*.
RESILIENCE	*Equivalent to adaptive capacity.*	Usually used as opposite of *vulnerability*. Based on: amount of change a system can absorb, ability of a system to self-organize, ability to increase capacity for learning and/or adaptation.	Related to ecological concept of a system's ability to return to initial state following perturbation. Acknowledges that a system may have multiple steady states.

These terms, by default, possess some ambiguity, as they are attempting to apply field-specific definitions to words that also have commonplace uses. This is compounded by the various subtleties in interpretation/application common in the field. In fact, to a reader outside the risk assessment community, many of the terms appear synonymous (e.g., *vulnerability* and *risk*; or *vulnerability* and *sensitivity*; Gallopín (2006) dedicates substantial discussion to differentiating *vulnerability*, *resilience* and *adaptive capacity*). Even the physical/geographic/societal unit(s) being assessed for risk are called by different names in different frameworks (e.g., 'elements,' 'system,' 'location' or 'community'). The following summary will use 'element' as it is the most general (if not most used) term.

In general, the terms are related in the following manner: *risk* is derived from the probability of an event (i.e., *hazard*) occurring, and the level of resultant impact (Turner et al., 2003). In everyday language, 'hazard' is usually associated with sudden dramatic events. However, the *hazards* considered in risk assessments related to climate change are often more gradual, continuous changes. Turner et al. (2003), distinguish these two types of *hazard* as 'perturbations' and 'stress/stressors,' respectively.

In order for a *hazard* to have an impact (and consequently present a risk), the element under consideration must be *exposed* to the *hazard* (e.g., inland communities are not typically *exposed* to marine *hazards*). The level of impact is dependent on how *vulnerable* the element is to the specific *hazard*. Cardona et al. (2012) make the distinction between *vulnerability* and *exposure*, by stating that in order to actually be *vulnerable* an element must first be *exposed*. *Vulnerability* refers to aspects of an element which affect the impacts that the element will experience if it is *exposed* to a *hazard*.

Confusion can also arise in the distinction between *risk* itself and *vulnerability*. While many frameworks and assessments use *vulnerability* as a component of risk, others address vulnerability as the primary concern. The difference largely lies in *vulnerability* being the tangible aspect of risk. *Risk* includes the probability that the *hazard* will occur, whereas *vulnerability* is based on the *hazard* occurring.

Sensitivity refers to how susceptible the element is to the *hazard*. Often this relates to how reliant an element is on a resource, the access to which will be affected by the *hazard*.

The difference between *sensitivity* and *vulnerability* is subtle, and *sensitivity* is frequently considered to be a key driver of *vulnerability*.

Adaptive capacity is the ability of an element to respond to a *hazard*. There is substantial discussion of *capacity* in risk management literature, as this is the aspect that is often most easily acted upon to respond to risk. *Adaptive capacity* is also often included (with *sensitivity*) as a factor of *vulnerability* (i.e., an element with a strong *capacity* to respond to a *hazard* is less *vulnerable*). There is sometimes differentiation between *adaptive capacity* and *coping capacity*. In these cases *adaptive capacity* refers to the ability to take proactive action to reduce *vulnerability*; and *coping capacity* refers to reactive potential of an element following a *hazard* to limit the impact (Cardona et al., 2012).

For a more technical discussion on the varying definitions of these terms, Gallopín (2006) presents a detailed review of most of the key terms as used in global change literature. His review also highlights the lack of agreement on definitions for many of these terms, and concludes "there is a need to develop clear […] specifications of these concepts in the abstract, ecological and social senses that are mutually compatible" (Gallopín, 2006, p. 302). Due to the inconsistent definitions it becomes necessary to define the exact definitions being used at the outset of each assessment.

1.6 Research Statement

Climate change and OA are already having impacts on fisheries; these changes are expected to intensify through the century and beyond. The goal of this thesis is to assess how expected changes in Atlantic Canadian fisheries will potentially affect human communities in the region. Atlantic Canada has a uniquely high dependence on invertebrate fisheries and is therefore expected to be a region with a particularly high risk to OA driven changes in fisheries.

To investigate this hypothesis, a review of the current methods used to investigate socioeconomic effects of OA-driven changes on fisheries was conducted (Chapter 2). Knowledge gained from this review was used to construct a risk framework for assessing the relative risk to Atlantic Canadian provinces from OA and climate change using modelled future distributions of seven highly valuable shellfish species (Chapter 3).

Chapter 2 Literature Review of Social and Economic Assessments of Ocean Acidification Impacts on Fisheries

2.1 Introduction

The ocean provides a vast array of ecosystem services. It has direct value for societies as a source of food and livelihood, and also holds significant social and cultural importance. Furthermore, it has absorbed over a quarter of anthropogenic CO_2 emissions since the Industrial Revolution (Hoegh-Guldberg et al., 2014; IPCC, 2014). Ocean acidification is a direct consequence of increasing atmospheric CO_2. Although, until relatively recently the potential impacts on biology were largely unrecognised (Doney et al., 2009; Orr et al., 2005); and climate change effects related to temperature have featured more prominently in both scientific literature and public awareness (IPCC, 2014). Following the recognition of OA and its potential effects on marine life (largely becoming mainstream beginning with the work of Orr et al. (2005)) research on the potential effects of OA, including studies focused on the potential costs to society, has expanded rapidly. However, studies specifically focusing on the human costs of OA are still limited. This may, in part, follow from biological research being more readily available for species that are not directly economically relevant, although these species may be indirectly important for economies if changes in their abundance or function alter ecosystem stability or structure. This is further confounded by closely related species that have, at times, exhibited markedly different responses to OA (Kroeker et al., 2013, 2010). Nonetheless, various authors have repeatedly recommended that despite the limited available data on biological effects, an understanding of potential socioeconomic impacts from OA and climate change should be developed (e.g., Cooley & Doney, 2009; Hilmi et al., 2013; Hoegh-Guldberg et al., 2014; IPCC, 2014). Hilmi et al. (2013) further suggest that this endeavour would be greatly advanced through interdisciplinary work, with natural scientists and economists working cooperatively, and acknowledging the strengths and weaknesses of both fields.

In a formal literature review targeting studies which consider the economic effects of OA, Falkenburg and Tubb (2017) identified 105 published studies that mention the potential for OA to lead to economic impacts. However, only eight of these specifically addressed and attempted to explicitly quantify potential economic impacts stemming from OA. Of these, five focused on invertebrate fisheries (Cooley & Doney, 2009; Lam et al., 2014; Moore, 2015; Narita et al., 2012; Punt, Poljak, Dalton, & Foy, 2014); the remaining three considered corals (L. M. Brander et al., 2012), seagrass (Garrard & Beaumont, 2014) and a single species finfish fishery (Voss, Quaas, Schmidt, & Kapaun, 2015). The rest of the 105 studies identified by Falkenburg and Tubb implied or discussed a relationship between OA and socioeconomics without directly assessing the interaction. To identify relevant publications, Falkenburg and Tubb (2017) searched through a series of databases of peer reviewed literature and used search terms that were focused specifically on economic impacts and ocean acidification. The search parameters used by Falkenburg and Tubb (2017) overlooked a sub-section of social effects which are not necessarily reliant on economic value, such as employment, livelihood or nutritional value. In contrast, many of these social costs, which are not readily reduced to prices, are addressed in risk and/or vulnerability assessments. While a number of these assessments were identified by the Falkenburg and Tubb's secondary search (which pulled articles from the reference lists of the publications returned by the initial search – and consequently they were counted among the 105 total identified studies), the studies did not specifically assess 'economic effects,' which the authors defined as making an explicit estimate of the monetary cost of OA.

Socioeconomic impact studies conducted to date have generally fallen into two broad categories, each with advantages and disadvantages: a) economic analyses, which seek to apply a dollar value to the projected impacts of OA; and b) social vulnerability and/or risk assessments, which take a broader approach to assessing the human cost of OA by including other social factors such as the relative importance of fishing industries to community livelihoods and abilities of human communities to respond to future changes. The former category of studies include a range of economic modelling methods to determine how the value of a given fishery will potentially change in the future, while the latter use risk or vulnerability frameworks to identify communities which are more likely to be impacted by OA.

There has yet to be a review specifically including these broader social assessments of potential impacts from OA. This may, in part, be due to the array of potential methods and outputs inherent in risk and vulnerability assessments, and/or related to the difficulty of directly comparing the outcomes of different risk assessments. Nonetheless, identifying common patterns in the methodologies used thus far can be useful for considering key factors influencing susceptibility to OA. This understanding could also help to inform methodologies of future socioeconomic assessments and lead towards more inter-relatable findings through standardization of approaches.

Due to the (currently) limited data regarding the effects of OA on finfish species (Kroeker et al., 2013, 2010), socioeconomic studies thus far have typically focused on shellfish fisheries. Some studies have addressed finfish (i.e., Voss et al., 2015), but this has mainly been through assumed ecosystem/food-web impacts (e.g., Heinrich & Krause, 2016; Mathis et al., 2015). Furthermore, given the prevalence of observed impacts on molluscs compared to those on crustaceans (Kroeker et al., 2013, 2010), the majority of these studies prioritise mollusc fisheries (although, again, exceptions exist (e.g., Heinrich & Krause, 2016; Punt et al., 2014)).

This chapter will review the methods used in various socioeconomic assessments, focusing on studies that assess the impacts of OA on shellfish fisheries, rather than on broader ecosystem services (e.g., tourism revenue or coastal protection from coral reefs). Limiting the scope of the review to assessments of OA impacts on fisheries allows for the methods to be more easily compared. Furthermore, fisheries represent an ecosystem service which is more straightforward to quantify, and changes therein are easier to directly apply to social (and economic) systems; since most other ecosystem services have more abstract measures and indirect impacts on human societies.

As noted above there are two main methodological approaches that have been used to address the social and economic implications of OA. For clarity, in this chapter the studies which followed more explicit economic quantifications will be termed ‘economic analyses.’ Studies which used risk or vulnerability frameworks will be collectively called ‘social vulnerability and risk assessments’ (abbreviated to SVR assessments). Together these two approaches will be referred to as ‘socioeconomic assessments.’

2.1.1 A Brief Outline of Social Vulnerability and Risk Assessment Methodology

Before considering the recent practice of applying SVR assessment methodology to OA, a brief overview of broader SVR assessment methodology is warranted[3] (see also Chapter 1 – section 1.5). SVR assessments have traditionally been applied to understand the risks posed by single extreme events (e.g., tsunamis, earthquakes, etc.) (Cardona et al., 2012). More recently, however, there is increasing interest in applying SVR assessment methods to more gradual, long-term pressures such as those presented by climate change (e.g., sea level rise, and OA) (e.g., N. A. Marshall et al., 2009; Turner et al., 2003). SVR assessment methods provide a framework for investigating the risks posed to communities, regions or nations from an array of hazards.

Most SVR assessment frameworks are built on three to five major components, depending on the approach employed by various authors. However, between assessments there are subtle differences in the definitions of these terms and in their relationships to each other (Figure 5). The main operational difference between the two SVR assessment types (i.e., risk assessments and vulnerability assessments) relates to the inclusion of a specific component for the hazard being assessed (Figure 5).

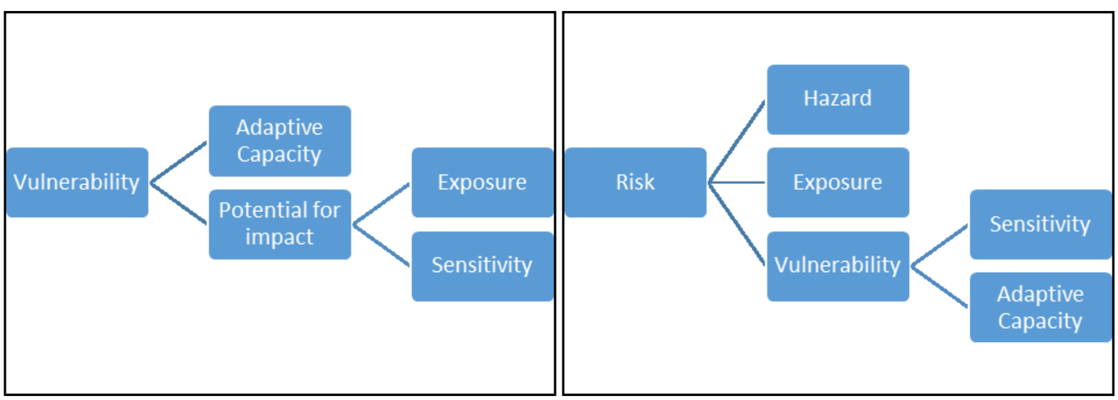

Figure 5. Example frameworks for vulnerability assessments (left – modified from (N. A. Marshall et al., 2009)) and risk assessments (right – modified from (Mathis et al., 2015)). Within assessment types variations exist regarding how links between components are arranged, as well as the relative importance of each component.

[3] Economic analyses are also discussed in this chapter, and while there is an enormous amount of theory behind the approaches followed in these types of studies the actual methodological steps followed are somewhat more direct than the frameworks involved in SVR assessment methods. As such a general overview of economic analysis theory was not included here.

The most common SVR framework components are variations on the following terms: 'hazard' describes the phenomenon under consideration that is expected to impact human systems in some way (Cardona et al., 2012; Turner et al., 2003). 'Exposure' relates to the elements or social units which will experience the hazard (Cardona et al., 2012; N. A. Marshall et al., 2009). The term 'vulnerability' usually describes how likely a social unit is to be affected by a hazard and itself is often represented as a combination of 'sensitivity' and 'adaptive capacity' (Cardona et al., 2012; Mathis et al., 2015). 'Sensitivity' indicates how dependent the social unit is on an affected resource (Cardona et al., 2012; N. A. Marshall et al., 2009). Lastly, 'adaptive capacity' typically indicates the ability of acommunity to respond to a hazard or change (Cardona et al., 2012; N. A. Marshall et al., 2009).

In risk assessments 'vulnerability' is a component of risk, while in vulnerability assessments, it is the main outcome of the assessment. In the latter, 'vulnerability' may also include 'exposure,' along with 'sensitivity' and 'adaptive capacity,' as a subcomponent (Figure 5).

Due to the unique dimensions of risk and vulnerability presented to specific geographies and social structures by different hazards, there is no single framework that fits all scenarios. Nonetheless, there is abundant literature regarding general outlines and considerations that should be followed when constructing an assessment framework (e.g., Cardona et al., 2012; Hajkowicz, 2006; Hobday et al., 2011; N. A. Marshall et al., 2009; OECD, 2008).

2.2 Methods

2.2.1 Literature Identification

Peer reviewed publications available online before December, 2016, which addressed social and/or economic impacts related to ocean acidification were identified. Google Scholar was used as the preliminary search tool for identification of articles, and reference lists of returned articles were consulted to identify additional publications. Primary search terms used to identify literature were 'ocean acidification,' 'socioeconomic,' 'fisheries' and 'economic' along with typical variations of some of these terms. A news aggregating website which reports links to new peer reviewed journal publications and media coverage related to OA was regularly consulted to identify newly published relevant articles throughout the research time-period [https://news-oceanacidification-icc.org/].

2.2.2 Thematic Analysis

The most intuitive division of identified assessments was between the economic analyses and the SVR assessments. Within these categories studies were evaluated to identify common themes and characteristics, which were used to further divide the studies and compare the various methodologies. A commonality in the methods of all of the studies was to first address and define the expected biophysical impacts from OA before assessing the impacts on social structures. These two general steps were common to all of the assessments and were examined sequentially for this review. Patterns in the biophysical components were considered first. Then, the tools used for assessing the social impacts were compared and themes were identified. Further similarities (e.g., geography and treatment of aquaculture) were also explored; however, these characteristics were generally independent from the identified organisational themes.

2.3 Results

Due to the relatively recent identification of OA as a potential challenge facing human societies (Doney et al., 2009; Orr et al., 2005), the body of literature addressing the potential social impacts related to OA is small. A total of 11 peer reviewed studies were identified which specifically assess how an OA-driven change in shellfish fisheries might affect human economies and communities[4]. A clear delineation was apparent between two broad methodological approaches of assessing the socioeconomic impacts of OA. Some studies addressed economic impacts, and others considered a broader social impact and used a form of risk or vulnerability assessment method to do so. The economic analyses specifically set out to understand how the economic value of the fisheries under consideration could be affected by OA. The SVR assessments attempt to address a broader social context and include factors related to the importance of the fisheries to local economies, and the ability of the social system to respond to potential changes.

[4] Two other publications completed portions of a socioeconomic assessment. However, their methods and findings did not formally define or quantify the potential social impacts from OA, and as such they were not included in the analysis (Hilmi et al., 2014; Kibria, Haroon, & Nugegoda, 2017)

These two main methodological groups were further divided into sub-categories defined by characteristics of their scope and specific approach (Table 2).

Table 2. Hierarchical framework of categories used to assess patterns between studies. Dashed vertical lines denote new subdivisions of the superseding category. The number of studies in each category is indicated in brackets. The final row includes the references for the finest scale categorisations.

Assessment type	Economic analysis **(7)**					Social vulnerability and risk (SVR) assessment **(4)**	
Biophysical-OA metric/ impact	Landings change proportional to change in pH **(4)**			Biophysical model **(3)**		Ω change + fisheries dependence **(4)**	
Biophysical-species selection	Mollusc **(3)**		Specific mollusc species **(1)**	Multiple species **(1)**	Single species **(2)**	Mollusc **(2)**	Shellfish + predators **(2)**
Social/ economic analysis tool	Value change proportional to change in landings **(1)**	Partial equilibrium supply/ demand model **(2)**	Perishable goods and social welfare **(1)**	Economic factors **(1)**	Integrated with bio-physical model **(2)**	Vulnerability assessment **(2)**	Risk assessment **(2)**
References	*(Cooley & Doney, 2009)*	*(Narita & Rehdanz, 2016; Narita et al., 2012)*	*(Moore, 2015)*	*(Lam et al., 2014)*	*(Cooley et al., 2015; Punt et al., 2014)*	*(Cooley et al., 2012; Ekstrom et al., 2015)*	*(Heinrich & Krause, 2016; Mathis et al., 2015)*

2.3.1 Geographic Ranges

Before examining the main methodological patterns between assessments, the following presents a brief summary of the various geographies considered in these studies to help frame the scales they consider. Geographical scopes used by the studies appear to be driven by one or more of three factors. First, the region being assessed is expected to experience stronger/more rapid OA than other regions (Heinrich & Krause, 2016; Lam et al., 2014; Mathis et al., 2015; Punt et al., 2014). Second, a shellfish fishery (i.e., a fishery expected to be susceptible to OA) is a significant part of the regional fishery and/or economy (Cooley et al., 2015; Heinrich & Krause, 2016; Narita & Rehdanz, 2016). Lastly some studies used a large geographic scope (including global-scale analysis), in order to arrive at results with high level relevance and to begin to consider the possible total extent of damages caused by OA (Cooley & Doney, 2009; Cooley et al., 2012; Ekstrom et al., 2015; Narita & Rehdanz, 2016; Narita et al., 2012). A notable similarity across most of the finer scale studies was a focus on developed nations or regions (Cooley & Doney, 2009; Ekstrom et al., 2015; Heinrich & Krause, 2016; Lam et al., 2014; Mathis et al., 2015; Narita & Rehdanz, 2016; Punt et al., 2014). This contrasts with the priorities outlined in the global scale vulnerability assessment (Cooley et al., 2012) which identified developing nations as being most susceptible to changes driven by OA. The greater focus on developed nations may, in part, relate to the first two factors (i.e., stronger OA and economically important shellfish fisheries in these regions) or overall data quality and availability.

2.3.2 Economic Analyses of Ocean Acidification

Seven of the identified studies fell into the economic analysis category (Table 2 – Row 1). The methods and terms used in these studies varied, but they were all centered on estimating a change in dollar value or revenue within a fishery or fisheries. In all cases, the biophysical factor had two distinct sub-components: a) the specific application of OA impacts to drive a change in the fisheries (Table 2 – Row 2), and b) the types of fisheries being assessed (Table 2 – Row 3). While the latter was more central to the study motivation, the former is useful to examine first as it facilitates sorting studies into more coherent groups. There were two main approaches used to model the impact of OA on fisheries landings. The first was a 'simple' application of experimentally observed changes in a life-history trait (such as growth or survival) to a change in fisheries landings (Table 2). Alternatively, the three remaining economic analyses applied a more complex biophysical model to estimate the effects of OA (Table 2).

2.3.2.1 Simple Modeling of Ocean Acidification Impacts

Choice of species: Four studies directly applied OA life-history effects to changes in landings of mollusc fisheries. Three of these assessed all mollusc landings in their study areas, while the fourth considered four specific mollusc fisheries (Table 2). In their assessment of American fisheries, Cooley and Doney (2009) applied the impacts from an early study of the effects of OA on two commercially relevant shellfish (Pacific oyster and Blue mussel) to all commercially harvested molluscs. Similarly, Narita et al. (2012) and Narita and Rehdanz (2016) applied results from a meta-analysis of projected impacts of OA (Kroeker et al., 2013, 2010) across a range of species to global (Narita et al., 2012) and European (Narita & Rehdanz, 2016) commercial mollusc harvests. The differentiation in underlying data likely reflects data availability at the time the studies were conducted.

The fourth simple model study assessed four fisheries (oyster, scallop, clam and mussel) in the United States (Moore, 2015), and based their species-specific impacts on results from a single publication which reported experimental observations of OA effects on 18 species of mollusc (Ries, Cohen, & McCorkle, 2009). The rationale provided by Moore for relying on this single laboratory study was that it included impacts for economically relevant species. Additionally, the research for this publication was actually conducted several years earlier (Moore, 2011), and at that time the available biophysical response data was limited compared to the more recent studies.

Modelling of OA impacts: Cooley & Doney (2009) set laboratory observations of decreased calcification for a given pH change, from Gazeau et al. (2007), as directly proportional to a change in landings. The change in pH for relevant ocean regions, informed by climate models, was then used to estimate how catches of molluscs may change for the United States. Narita et al. (2012) followed the methods of Cooley & Doney (2009) for the OA effect, but used data from a meta-analysis of biologic responses (Kroeker et al., 2010) to inform the OA effect. Additionally, they compared outcomes for two different physiological impacts: reduced calcification and reduced survival, although they focused their discussion on the former. Narita and Rehdanz (2016) continued the approach of applying the physiological impacts directly to fishery landings, but they used an updated version of the meta-analysis (Kroeker et al., 2013) to inform their impact levels. Moore (2015) similarly used a measure of life-history impact – in this case change in growth rate – due to OA to directly project a change in fisheries landings.

Estimates of economic impact: While the studies discussed thus far fall into the 'economic analysis' category, there were notable differences in the methods applied to arrive at the economic outcomes. Cooley and Doney (2009) used their projected change in fisheries landings to proportionally change the value of the fisheries in the future. Present value of the future losses were estimated using a range of discount rates while all other economic factors were assumed to be static.

While they followed Cooley and Doney (2009) for the biophysical component of their analysis, Narita et al. (2012) diverged from the earlier study in their economic methods. A partial-equilibrium model (which uses supply and demand relationships to estimate how a change in supply will result in a change in value for a resource) was used to estimate the value of future losses in harvest, in place of making economic losses directly proportional to fisheries losses. The partial-equilibrium model allowed for anticipated declines in supply to affect the predicted value of the fisheries. They suggested that future work could go even further and use a 'general-equilibrium' model that would allow for changes in shellfish production to affect global trade and production in other markets. Narita & Rehdanz (2016) closely followed Narita et al. (2012) in their approach but for a smaller geography (Europe) and with some minor adjustments to the partial-equilibrium model. They also compared the results of two methods of economic analysis: a simple relation between the OA impact and the economic loss, as in Cooley & Doney (2009), and contrasted this with the partial-equilibrium model as used in their earlier study (Narita et al., 2012).

Moore (2015) pursued a more specific economic assessment and estimated the welfare costs to consumers resulting from an OA driven reduction in mollusc harvest in the United States. By following economic assumptions intended for perishable goods (including seafoods; (Barten & Bettendorf, 1989 in; Moore, 2015)) the study design emphasised how changes in supply of the respective shellfish species could potentially affect consumer welfare. Including species-specific impacts allowed changes in each individual species to affect demand for the other assessed species.

2.3.2.2 Complex Biophysical Modelling of Ocean Acidification Impacts

The three remaining economic analyses employed more complex biological models to determine how target species populations might change in response to OA, and how these changes might affect future landings (Table 2). Cooley et al. (2015) and Punt et al. (2014) both took targeted approaches and used models built for specific high value, single species fisheries and included explicit OA effects for multiple life-stages. Lam et al. (2014) used a DBEM with more generalised OA impacts derived from a meta-analysis (Kroeker et al., 2013, 2010) to model effects of OA on growth and survival. The model used by Lam et al. (2014) was initially built to predict climate change-driven species distribution shifts, and therefore included climate change factors beyond OA (Cheung, Lam, & Pauly, 2008).

Single species models: Both of the studies assessing single species fisheries used models that integrated landings predictions with the biophysical changes, so that the OA effects and the change in value of the fishery were directly linked. Punt *et al.* (2014) integrated two biological models for red king crab (*Paralithodes camtschaticus*) in Alaska: a) a 'pre-recruit model' which predicted recruitment from larval stages to the adult stage, and b) a 'post-recruit model' which calculated the adult (harvestable) population (Figure 6). The OA effects were based on two studies which specifically examined developmental stages for red king crab (Long, Swiney, & Foy, 2013; Long, Swiney, Harris, Page, & Foy, 2013). The adult (post-recruit) model did not explicitly include an OA term, and was instead influenced by population effects carried over from the OA impacts in the larval model. The post-recruit model outputs were used to predict changes in catch rates for the fishery. The costs associated with fishing activity were calculated into the future and were combined with the projected changes in landings to determine how profitability in the fishery could change (Figure 6). The use of the species-specific OA impacts allowed this study to present a detailed assessment of how the red king crab fishery might change in response to OA. Nonetheless, the authors noted that the study was restricted by the available biological data and the limited number of published studies for their species.

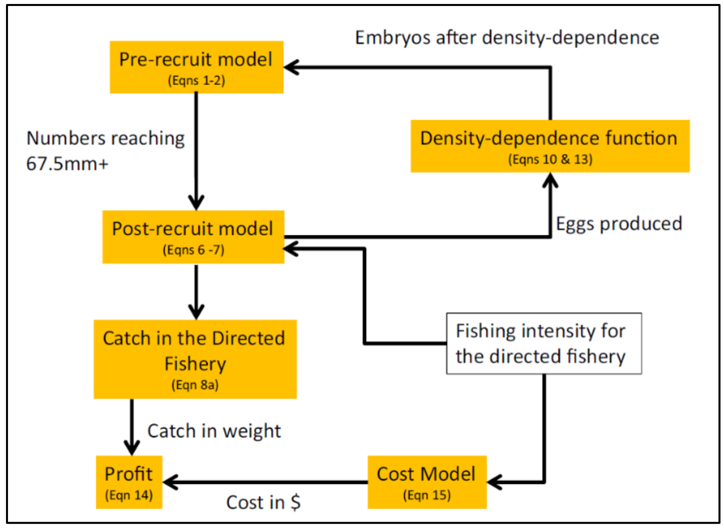

Figure 6. Framework for integration of biologic and economic models in Punt et al. (2014, p. 40).

The second species-specific study to use a more complex biophysical model of OA impacts focused on the sea scallop (*Placopecten magellanicus*) fishery in the Northeastern United States (Cooley et al., 2015). In this study three separate models were linked together to determine the economic impact of OA on the fishery. A geochemical sub-model, with surface and deep ocean components, was used to predict changes in ocean chemistry. Outputs from this model were used to drive changes in scallop recruitment, growth and mortality. The scallop population sub-model in turn provided an input to the socioeconomic sub-model; the socioeconomic sub-model also included a feed-back to the scallop sub-model by affecting adult population and hence available biomass for the fishery (Figure 7).

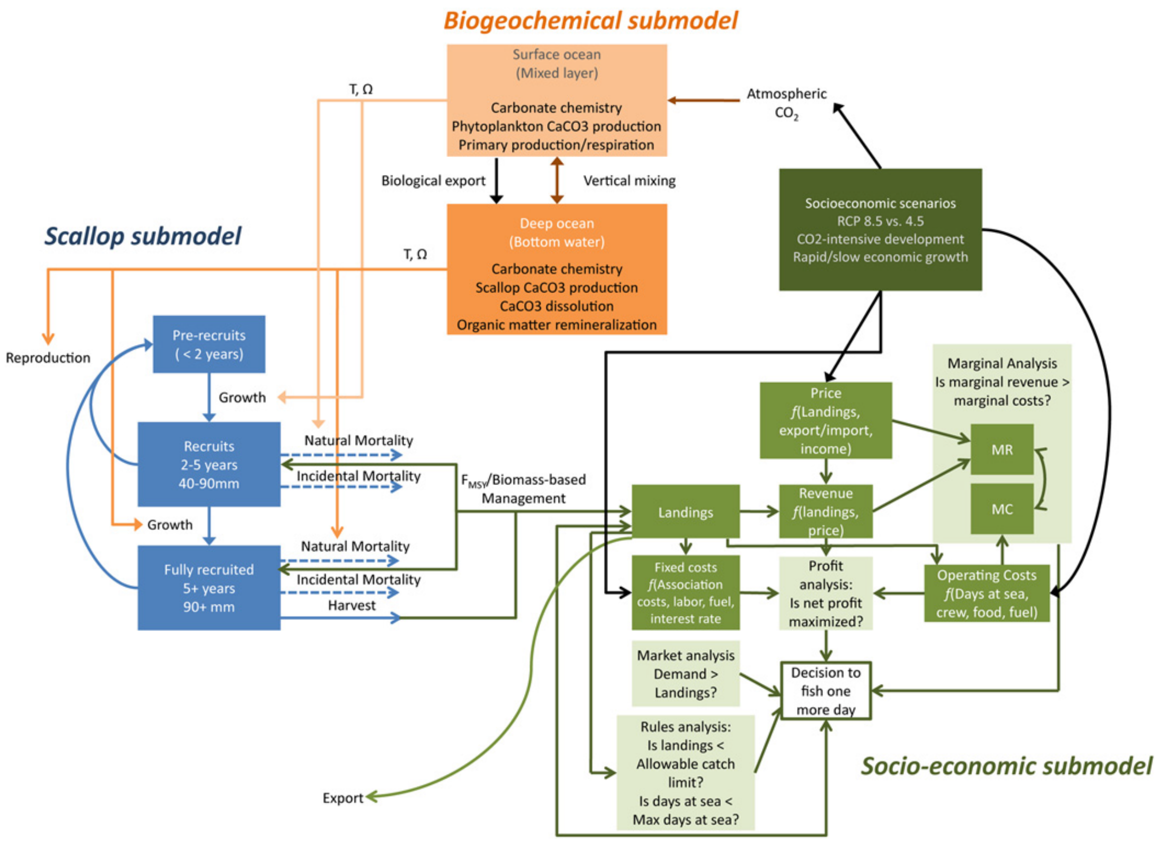

Figure 7. Integrated assessment model schematic from Cooley et al. (2015, p. 5).

At the time of the study no species-specific data were available for adult sea scallop growth or survival under future OA conditions, so the authors conducted a meta-analysis of available biological OA impact publications for other bivalve species to obtain an approximation of the impact on adult sea scallops to use in the model. The larval survival was based on a separate laboratory study for a closely related scallop species with similar life-history characteristics (Andersen, Grefsrud, & Harboe, 2013). The socioeconomic sub-model also included factors associated with fishing such as operational costs and catch value to ultimately assess whether the value of the fishery economically justified the effort and costs to operate (Figure 7).

A noted advantage of the modular construction of the integrated model was the option to easily update the sub-models as new scientific data becomes available. Similar to the analysis by Punt et al. (2014), the more detailed modelling of the life-cycle stages for the target species allowed for a more precise estimate of the economic impacts to the fishery.

Multi-species biophysical model: Lam et al. (2014) addressed 62 commercially harvested species in the Arctic ocean; however, their OA impacts were not species-specific. Instead the Kroeker et al. (2013, 2010) meta-analyses impacts were applied by taxonomic group, with mollusc and crustacean species receiving separate impacts on growth and survival, while finfish species were assumed not to be affected by OA. This study used a DBEM built by Cheung et al. (2008) to predict future changes in landings as a result of climate change effects. The DBEM uses current species distributions to identify suitable habitat conditions (e.g., temperature and depth), and then predicts where suitable habitat will exist in the future using global climate models. This is linked with a series of additional models, including species distribution and population dynamics models in order to predict future distributions and abundances (see Chapter 1, section 1.4 for a summary of the DBEM model). For their analysis, Lam et al. (2014) incorporated OA into the DBEM (building from earlier work by Cheung et al. (2011)) as an effect on growth and survival of mollusc and crustacean species groups. The DBEM also includes temperature effects on abundance and distribution. This is particularly noteworthy as changes in temperature are likely to have overall greater potential impact on survival and abundance than OA (Portner, 2012). In order to isolate the OA impact on potential harvests, the DBEM was operated with and without the OA parameters, and the trials were compared against each other. The economic consequences were not integrated with the biophysical model, instead Lam et al. (2014) considered five different metrics of economic value (total value of fisheries, income from fisheries, costs associated with fishing economic rents, and impacts on wider economies), which were affected by the projected changes in harvest.

2.3.3 Risk and Vulnerability Assessments of Ocean Acidification

Four studies of potential social impacts resulting from OA fell into the domain of SVR assessments (Table 2 – Row 1). All SVR assessments inherently include both biophysical and socioeconomic dimensions. The influence or impact of OA filled the role of the 'hazard' or 'exposure' term in the assessment frameworks (Table 3). However, the actual representation of OA varied considerably between studies. Similarly, the social and economic components of the assessments addressed related themes but were built for different social contexts and data. The specific indicators applied were generally selected based on their relevance to the social system being assessed, but were also at times limited by available data (Ekstrom et al., 2015; Heinrich & Krause, 2016).

The SVR assessments, by definition, had two sub-groups: a) risk assessments (Heinrich & Krause, 2016; Mathis et al., 2015), and b) vulnerability assessments (Cooley et al., 2012; Ekstrom et al., 2015) (Table 2 – Row 4). The main operational difference between the framework designs was the inclusion of a hazard term in the risk assessments. The inclusion of the extra term (or lack thereof) also forced some related differences in the orientation of the other framework components (e.g., where OA actually fits into the framework) (Figure 5; Table 3).

Table 3. Social vulnerability and risk assessment framework components and indicators as defined and measured in each study. Some studies combine components in different orders and/or include/exclude components. Vulnerability assessments did not include an explicit hazard term. These studies incorporated exposure, sensitivity and adaptive capacity under the vulnerability index. The risk assessment frameworks defined risk as the culmination of hazard, exposure and vulnerability; with the latter being represented as a combination of sensitivity and adaptive capacity. Adaptive capacity was included in one of two ways either to: a) reduce vulnerability (or risk), and it was counted against the overall indexed score, or b) a lack of adaptive capacity was used to add to vulnerability (or risk). When specific variables were not defined for a given framework component or when the variables used are not fully consistent with expectations or other studies, an explanation is presented in italics.

Study	Hazard	Exposure	Sensitivity	Adaptive Capacity
(Cooley et al., 2012)	*Not explicitly considered in the framework*	Transition decade – when annual variation in Ω no longer overlaps with current variation	-Mollusc contribution to GDP -Mollusc contribution to protein intake -Dietary protein deficiency	*Focused on lack of adaptive capacity (i.e. items contribute to vulnerability)* -Production increase required to maintain current consumption -Absence of aquaculture -Low average adaptability score (GDP, governance, literacy and life expectancy)
(Mathis et al., 2015)	Ranked ocean regions by change in mean Ω by 2090-2099	-Fraction of total fisheries value from shellfish and salmon (salmon weighted half as heavily as shellfish) -Fraction of subsistence harvest from shellfish and salmon (salmon weighted half as heavily as shellfish)	-Revenue per capita from harvesting and processing of shellfish and salmon -Per capita sustenance harvest weights of shellfish and salmon	-Economic stability (combination of: average income, dependence on government funding, poverty, and unemployment) -Education -Alternative employment -Food accessibility
(Ekstrom et al., 2015)	*Not explicitly considered in the framework*	-Time until Ω = 1.5 -Local amplifiers (freshwater inputs, upwelling, and eutrophication)	-Value of shellfish harvest -Contribution of shellfish to overall fisheries value -Number of shellfish licences	-Existing policy for OA and climate change -Alternative employment options -Scientific capacity
(Heinrich & Krause, 2016)	Ranked ocean regions by change in Ω	Value of harvest for defined species groups	Income from fishing compared to average income	-Personal income -Poverty -Unemployment -Education

2.3.3.1 Implementation of Ocean Acidification

The four SVR assessments all addressed the level of expected OA in the future as an independent factor from fisheries landings. Cooley et al. (2012) undertook an assessment at a global level and ranked nations by the time until annual maximum average Ω becomes less than current annual average minimum Ω. Mathis et al. (2015) and Heinrich and Krause (2016) compared anticipated change in Ω from present values for ocean sub-regions in their assessment areas (Alaska and Norway, respectively); the ocean areas were subsequently ranked relative to each other (Table 3). In the most nuanced representation of OA, Ekstrom et al. (2015), based the OA term in their framework on the time until Ω falls below 1.5 in adjacent marine 'bioregions' based on oceanographic models. Importantly, this was combined with local factors which affect pH in order to account for the significant complications that arise when projecting OA in coastal environments (Table 3) (e.g., Doney, 2010; Duarte et al., 2013).

Within the assessment frameworks, the OA projections were represented under one of two framework components. Mathis et al. (2015) along with Heinrich and Krause (2016) (i.e., the risk assessments) used the change in OA to represent the hazard underlying the assessment (Table 3). Alternatively, Cooley et al. (2012) and Ekstrom et al. (2015) (i.e., the vulnerability assessments) used the change in Ω to define exposure of their social units to OA (Table 3). This differentiation establishes the main divide between the risk and vulnerability assessments, and cascades through the orientation of the indicators in the rest of the studies' frameworks. Since the risk assessments included OA as a separate component of their frameworks, the other components (including exposure) were directed more specifically towards the social structure and how OA may affect the society. Conversely, the vulnerability assessments included the OA factor in their exposure component, and concentrated the social indicators under the sensitivity and adaptive capacity components.

2.3.3.2 Species Selection and Socioeconomic Indicators

Fisheries selection and inclusion within frameworks: As in the economic analyses, the SVR assessments in this review largely focused on shellfish fisheries. Two of the studies focused exclusively on mollusc fisheries, one at a global scale and the other for all of the United States (Table 3) (Cooley et al., 2012; Ekstrom et al., 2015, respectively). The remaining assessments considered impacts on shellfish fisheries more generally. This decision was likely influenced by the importance of crustacean species among shellfish landings in respective study areas. Mathis et al. (2015) evaluated OA impacts on fisheries in Alaskan counties and Heinrich & Krause (2016) examined impacts on Norwegian counties. Notably, this pair of studies also attempted to account for impacts on selected finfish which prey on OA susceptible species. However, in both instances, the direct impacts on shellfish were weighted more heavily than the indirect impacts on finfish (Table 3) (Heinrich & Krause, 2016; Mathis et al., 2015).

Economic indicators: In contrast to the economic analyses, which focused on estimating specific changes in landings and associated values, the SVR assessments considered the importance of susceptible fisheries to the relevant social units that are supported by the fisheries. Nevertheless, the SVR assessments included a range of economic indicators to assess how potential changes in future fisheries could affect local economies. Given the flexibility inherent in SVR framework construction (Turner et al., 2003), it is not surprising that the OA risk and vulnerability assessments have used socioeconomic indicators very differently when outlining their frameworks. Both of the risk assessments (i.e., Heinrich & Krause, 2016; Mathis et al., 2015) used the overall value of fisheries as an indicator of the exposure of counties to OA; they also addressed the per capita importance of the fisheries but scored this under the sensitivity term of their frameworks. In contrast, Cooley et al. (2012) and Ekstrom et al. (2015) used the fisheries' economic (and nutritional, in Cooley et al. (2012)) importance to represent the sensitivity of the social system to OA (Table 3). The differences between these frameworks demonstrates the versatility of the SVR assessment approach: substituting indicators that are more relevant to the geography and society being assessed allows for case-specific identification of most socially vulnerable/at risk communities.

Other social indicators: Beyond the economic indicators, the SVR assessments incorporated additional social indicators to quantify the overall importance of relevant fisheries to the social structures being assessed. The four SVR assessments diverged most widely from each other with respect to the specific social indicators they incorporated into their frameworks. These indicators were distributed between the sensitivity and adaptive capacity components of the frameworks (Figure 5). The sensitivity indicators tie the society to the assessed fishery; while adaptive capacity represents the broader social context, with many of the contributing indicators reflecting a generally more stable and adaptable society.

Mathis et al. (2015) and Cooley et al. (2012) included terms representing the nutritional importance of species affected by OA (the portion of subsistence harvest derived from OA susceptible species; and the proportion of dietary protein from molluscs, respectively) to bolster the economic aspects of their sensitivity component (Table 3). Similarly, Ekstrom et al. (2015) included the number of fishing licences as a proxy for employment reliant on shellfish fisheries, to reinforce their definition of sensitivity (Table 3).

In all four studies, broader social measures were used to inform adaptive capacity. In general these indicators were not directly relevant to the fisheries themselves. The most common feature included under adaptive capacity was a metric representing alternative employment options and/or unemployment rates (Table 3) (Ekstrom et al., 2015; Heinrich & Krause, 2016; Mathis et al., 2015). Mathis et al. (2015) and Heinrich and Krause (2016) also included measures of general economic stability (including average incomes and poverty rates), as well as educational attainment levels (Table 3). Ekstrom et al. (2015) focused on government and policy aspects of adaptive capacity. Lastly, Cooley et al. (2012) included a more general adaptability score comprised of Gross Domestic Product (GDP), governance, literacy rates and life expectancy of the local community to inform their adaptive capacity component (Table 3). Notably, Cooley et al. (2012) was the only study to explicitly include (a lack of) aquaculture as an indicator in their framework (Table 3).

2.4 Discussion

2.4.1 Representation of Ocean Acidification

All of the assessments built from an estimate of the OA impact on biology. Interestingly, the economic analyses all estimated a direct OA effect on a species, or taxonomic group, and for the most part did this by representing OA as a change in pH. This likely follows from the tendency of laboratory experiments to present biological responses (e.g., changes in growth, survival or larval development) in relation to pH levels (Kroeker et al., 2013, 2010). Using a change in pH to represent OA allowed the experimentally derived impact to be directly applied to changes in harvest (Cooley & Doney, 2009; Lam et al., 2014; Narita & Rehdanz, 2016; Narita et al., 2012).

Cooley and Doney (2009) used the experimentally observed change in calcification for a given change in pH to estimate a projected change in landings. The authors acknowledged this was a simplified relationship, but, proposed that the study was primarily a starting point to spur discussion of potential impacts, and provided a benchmark against which to compare future studies. Furthermore, since the approach was straightforward it allowed for similar analyses to be conducted in other geographies, as seen in Narita and Rehdanz (2016), Narita et al. (2012) and Moore (2015). Some of these later studies used alternative physiological responses such as growth or survival, but followed Cooley and Doney (2009) in assuming these changes were directly proportional to changes in landings.

The studies that built from more complex biophysical models simulating fishery impacts included more mechanistically detailed applications of the OA effect to the species of concern (Cooley et al., 2015; Punt et al., 2014). These studies considered multiple aspects of a species' life-cycle and applied the OA effect to more susceptible life-stages, thereby, using potentially more representative OA impacts.

In contrast, the SVR assessments circumvented the need for an explicit modelled relationship between a level of OA and a specific change in landings by including the OA term and the importance of a target species as two separate indicators. The four SVR assessments considered here all estimated the amount of OA for their respective study areas through separate methods and metrics (Table 3). But in all cases this estimation was undertaken independent from their consideration of relevant fisheries. With respect to their choice of fisheries, two of the studies specifically considered the importance of mollusc species to the communities being assessed, as molluscs are expected to be most susceptible to OA (Cooley et al., 2012; Ekstrom et al., 2015). The other two studies allowed for different OA effects between species groups and weighted the contribution of these groups in their assessment score according to their expected (relative) susceptibilities (Heinrich & Krause, 2016; Mathis et al., 2015). This presented a somewhat wider view of the potential impact of OA to the fisheries sector as a whole, but also relied on less certain biological responses.

2.4.2 Inclusion of Other Stressors

It is generally recognised that biological responses to OA will be complicated by responses to other climate change stressors (e.g., changes in temperature, salinity, dissolved oxygen) (Byrne & Przeslawski, 2013; Kroeker et al., 2017; Portner, 2012). Given the complexity of the variables and their interactions, there is far less understanding of combined potential impacts than there is of the (already limited understanding) of biological effects of OA itself. It is not surprising then, that the majority of the existing socioeconomic studies considered the impact of OA in absence of other climate stressors. Alternatively, this may be a feature of the framing of the studies, which sought to specifically highlight the potential social impacts of OA in light of its relatively recent recognition as a threat to the marine resources.

However, the focus on OA-specific impacts does not present the whole picture. For example, changing temperature gradients may drive shifts in species distributions and increase regional abundances in spite of OA effects. Lam et al. (2014) built their assessment on an existing model designed to predict changes in species distributions in response to a variety of climate change stressors. Through this, they present an example of how OA effects can be modelled in combination with other aspects of climate change. Nevertheless, as with the other publications discussed here, Lam et al. still focused on OA-specific findings by isolating the OA effect from the other model factors. Due to the focus on Arctic fisheries, increasing temperatures were predicted to drive increased abundances of many commercially valuable species. When the OA effect was included, the temperature-driven increases were attenuated, but nonetheless most species still had net gains in abundance. Lam et al. (2014) also noted that the relatively minor impact of OA on changes in overall fisheries value was, in part, due to the comparatively small contribution of shellfish to fisheries in the region. As seen in this study, OA will likely negatively affect landings; however, the overall climate change (i.e., temperature-induced) effects have the potential to outweigh the losses, at least in some regions. The authors of the study acknowledged that while multiple climate change factors were included in their model, that the lack of scientific data on how these factors will interact suggests the conclusions could be conservative with respect to predicted outcomes, particularly in terms of OA impacts.

Given the general focus on developed nations in temperate to polar latitudes that is common in the socioeconomic assessments (i.e., Cooley & Doney, 2009; Ekstrom et al., 2015; Heinrich & Krause, 2016; Lam et al., 2014; Mathis et al., 2015; Narita & Rehdanz, 2016; Punt et al., 2014), it is possible that the many of regions studied so far may experience net increases in landings for many of the assessed fisheries (e.g., Cheung et al., 2010, 2011). While OA will likely diminish or even negate these gains, an OA-only focus may present conclusions that could encourage mitigation actions (including additional research) that might be better prioritised elsewhere.

Besides interactions between physical stressors on individual species, it is also probable that changes in ecosystem dynamics will indirectly result from OA (as well as other climate stressors) (Gaylord et al., 2015; Kroeker et al., 2017). This is also a complicated system to model accurately, especially given the limited OA response data upon which predictions would have to be based. Mathis et al. (2015) and Heinrich and Krause (2016) partially accounted for this by including an OA effect for finfish species that prey on OA-susceptible invertebrates. Although this was a very limited approach to accounting for broader ecosystem impacts, it does provide preliminary insight into how OA may affect fisheries beyond shellfish.

Taking ecosystem effects a step further, a recent publication from Marshall et al. (2017) presented an alternative biophysical model that has strong potential for informing future socioeconomic assessments of OA. A distinctive factor included in the Atlantis model (Fulton et al., 2004) is ecosystem effects and interactions – a concept that is almost completely absent from the other OA assessments conducted to date. A significant advantage of a biophysical model that includes ecosystem links is the ability to include species which are not directly affected by OA, but do depend on susceptible species as prey. Marshall et al. (2017) suggest that the model as a highly suitable input for future SVR investigations of the region[5].

[5]In their study, Marshall et al. (2017) discuss the economic impact to a certain degree, but are predominantly focused on the presentation of the ecosystem impacts for the California Current system. The application of their findings to economic value is similar to that of earlier economic assessments (i.e., Cooley & Doney, 2009) and makes the changes in economic value directly proportional to projected changes in landings.

2.4.3 Fisheries Selection and Underlying Data

A subset of the OA assessments reviewed here based their OA impacts on the results of individual experimental studies (Cooley & Doney, 2009; Moore, 2015). Cooley & Doney (2009) presented the first socioeconomic assessment of OA and conducted their study before much of the now-extant experimental research was available, let alone the meta-analyses by Kroeker et al. (2013, 2010). The biological data used by Moore (2015) (i.e., Ries et al., 2009) allowed species-specific impacts to be included, and this allowed for the differences between species responses to play a role in how the overall value of the industry might change. Nonetheless, as with Cooley & Doney (2009), the results and conclusions were ultimately reliant on the findings of a single species impact study, however robust or limited it may have been. The socioeconomic studies that incorporated results from meta-analyses (i.e., Kroeker et al., 2013, 2010) to define the OA impact may be grounded in more robust biological understanding of potential OA responses but with a trade off with respect to location or species-specific responses.

The two single-species studies also based their OA impacts on a limited number of experimental results; although, these were selected to be relevant to their respective fisheries (Cooley et al., 2015; Punt et al., 2014). Both of these studies focused on high-value fisheries, which had at least some species-specific OA impact data available to inform their models of potential future change in abundance. Although, Cooley et al. (2015) still had to conduct an internal meta-analysis of reported impacts for closely-related species in order to address data gaps for the OA impacts on adult sea scallop. These studies present a finer-resolution assessment of how specific fisheries may be affected, but are nonetheless limited by the small number of OA experiments currently available. As the understanding of species-specific OA impacts increases, these types of analyses will become much more robust.

Due to the large degree of uncertainty and variability in observed responses to OA many of the studies discussed here based their biological impacts on the results of the meta-analyses conducted by Kroeker et al. in 2010 and 2013. These meta-analyses derived ranges of impacts for broad taxonomic groups from a wide array of experimental studies. Overall, molluscs were found to be significantly negatively affected by OA. Crustacean species tended to be negatively impacted, although the cumulative experimental research findings were not statistically significant. Available studies on finfish as a whole indicated variable responses and had no significant trend towards either positive or negative effects (Kroeker et al., 2013, 2010). The ranges of responses to OA identified by Kroeker et al. have been implemented differently across studies of potential socioeconomic OA impacts. Some of the studies used the upper and lower limits of the impact ranges (e.g., Narita & Rehdanz, 2016), while other assessments based their analyses on the mean impact level (Lam et al., 2014; Narita et al., 2012). The mean impact may be more representative of the most likely scenario. Nonetheless, the assessments that consider the more extreme OA impacts do allow consideration of the 'worst case' scenarios for future fisheries.

Since the Kroeker et al. (2013, 2010) meta-analyses presented their findings for broad taxonomic groups, the subsequent socioeconomic assessments tended also to aggregate species impacts to the same taxonomic levels (i.e., molluscs and/or crustaceans). Consequently, the assessments grouped all mollusc fisheries in their study region under a single OA effect, regardless of actual species composition. Lam et al. (2014) partially compensated for this by applying the OA impact to each species separately and allowing it to act on their life-histories independently within their model. Nonetheless, the level of impact (i.e., the effects on growth rate and survival) was still the same across species within the taxonomic groupings.

All of the socioeconomic assessments acknowledged the high variability and inconsistencies in the biological response data as a source of weakness in their analyses. They further recommended additional research in the field, and promoted a focus on investigations of how OA will interact with other climate stressors such as temperature change.

2.4.4 Social Analyses and Methodologies

2.4.4.1 Economic Analyses

Building from their straightforward OA effect on future landings, Cooley and Doney (2009) undertook the simplest approach to estimating economic impact by applying the predicted loss in landings directly onto the economic value of the fisheries in their study.

Subsequent studies employed more technical economic methods in attempting to arrive at potentially more representative potential future economic impacts. Narita and Rehdanz (2016) compared their supply and demand model against a proportional loss in value (*sensu* Cooley and Doney, 2009), and determined that the more complex model yielded smaller economic losses since to the model accounted for changes in value resulting from changes in demand as supply decreased. Moore (2015) also presented a more complex economic analysis. The economic analysis used in that study, along with the species-specific biophysical impacts, sought to predict how OA might affect consumers in the United States. By considering shellfish to be a perishable good, Moore (2015) framed OA in a context that is most relevant to developed nations. This style of economic analysis would not be as applicable at a global scale or in a context where shellfish represent an essential dietary component.

2.4.4.2 Social Vulnerability and Risk Assessments

All of the SVR assessment studies reviewed here accomplished their social analysis with the use of a framework wherein the social units being assessed were scored on each component and ranked against each other to determine where the highest (and lowest) relative risk or vulnerability occur. The ranking is particularly useful for highlighting regions which might be more immediately affected by OA. However, the scores and ranks are only relative to the other units in the study. If a fishery considered by a study turns out to be highly affected by OA, it is possible that even the lowest risk areas could be highly impacted (conversely, if the effects of OA prove to be minimal for the target species, the highest risk areas might experience little impact). Additionally, since the suite of indicators was different for each study, studies conducted for a similar region, or at a similar resolution, are not directly comparable.

Nonetheless, there is still some utility that can be derived from conducting separate risk assessments. Studies conducted for the same geography could reinforce or challenge earlier findings regardless of specific indicators and frameworks. In this sense, if multiple studies determine that overlapping regions are at higher risk, even when different indicators are considered, this would strengthen the conclusions and could be used to further prioritise mitigating action. Alternatively, if findings contradict each other there is still a potential for advancement through identifying which indicators are most significant drivers of OA-related risk (Turner et al., 2003). If, for example, high unemployment was a consistent indicator in high risk/vulnerability communities across studies, which also considered other unique indicators, unemployment could be targeted as a factor to reduce overall susceptibility to OA.

2.4.5 Consideration of Aquaculture

Globally, aquaculture production is growing while capture fisheries landings remain relatively stable. Thus, aquaculture is expected to play an increasingly important role in future food security (FAO, 2016). It is, therefore, difficult to ignore aquaculture in discussions about the future of fisheries and seafood production. Aquaculture is especially relevant in the context of shellfish production, since marine aquaculture is dominated by shellfish (15.8 million T globally in 2014) and represents more than double the finfish production (6.3 million T) (FAO, 2016). Wild harvest is primarily derived from finfish fisheries (e.g., Pauly & Zeller, 2015). Furthermore, given the breadth of interventions possible in aquaculture, from rearing juveniles under ideal conditions to selection for specific traits, there is potential for aquaculture to be more resilient to OA than wild harvest (Clements & Chopin, 2016). Consequently, aquaculture should be an important feature in discussions of future shellfish production as it relates to OA. Despite the clear role that culture can play, it is interesting that to date, aquaculture has not played a role in the majority of socioeconomic assessments of OA impacts. Although, this may be, in part, affected by data availability rather than active decisions to exclude aquaculture production.

Some of the economic analyses acknowledged that aquaculture operations may have the capacity to respond to OA, especially where hatcheries are used for rearing through larval stages[6] (Narita & Rehdanz, 2016; Narita et al., 2012). However, these studies also suggested that because shellfish aquaculture (i.e., filter feeding organisms which largely rely on the marine environment for food inputs) is still dependent on ocean resources for growth, and given that impacts remain uncertain, the most conservative assumption was

[6] As opposed to situations where aquaculture production relies on wild populations or natural breeding of cultured organisms for recruitment.

to combine aquaculture and capture production statistics (Narita & Rehdanz, 2016; Narita et al., 2012). A potential alternative to this assumption could have been to explicitly incorporate aquaculture into the analysis through a modified or reduced OA impact on aquaculture production (e.g., applying a fraction of the OA effect to aquaculture production). Another alternative would be that aquaculture could be entirely, and explicitly, excluded from the assessment, and capture fisheries could be made the core focus of the assessment, as performed by Lam et al. (2014).

The SVR assessment approach allows for more options to potentially account for aquaculture interventions/mitigation in response to changes in future shellfish production. However, the framework designs used in the studies reviewed here mainly only assessed wild harvests or treated aquaculture production the same as wild harvest in their exposure terms. In some cases this may have been related to the inclusion of aquaculture posing an additional complication when consumption/nutrition statistics were included (e.g., Cooley et al., 2012; Mathis et al., 2015). The underlying data for consumption statistics (in most cases) are unlikely to differentiate between sources of seafood production, this would present a disconnect within the assessment framework, and lead to the resource being assessed being treated differently in the exposure and sensitivity components (i.e., a portion of the sensitivity score may be based on a resource (aquaculture) that is not, in fact, fully exposed to the hazard).

Future SVR frameworks could potentially work to account for aquaculture resilience to OA by explicitly including aquaculture production in the exposure term and differentiating its treatment from capture fisheries. This approach could be applied in a similar manner to how Mathis et al. (2015) included exposure terms for both shellfish and finfish but with different weights in the assessment framework. Since adaptive capacity does not directly relate to the hazard itself, the adaptive capacity component could also be used to highlight the potential for aquaculture to respond to OA and reduce the overall risk or vulnerability in an area, as seen in Cooley et al. (2012). Care should be taken to ensure that the manner in which aquaculture is included under adaptive capacity does not conflict with how it is handled in the other framework components. If resistance of aquaculture to OA is accounted for under exposure, also including it under adaptive capacity could lead to over-estimating the impact that culture could have for limiting risk and vulnerability to OA. Furthermore, to include aquaculture in a framework, it must be a viable option for the study area. If aquaculture would not be economically feasible, or the species being considered were not suitable to culture, it would not be reasonable to include it as an indicator of adaptive capacity (or any framework component).

2.5 Literature Review Conclusion

2.5.1 Economic Analyses versus Risk Assessments

The two main socioeconomic assessment categories both have advantages and disadvantages with respect to their methodologies and application of their findings. Economic analyses have the advantage of returning results which have a consistent, familiar and, in theory, comparable unit of measure between studies. Although, Brander et al. (2014) note that the inconsistent methodologies and assumptions in the current literature mean that results of the available economic impact studies of OA are not directly comparable. Nonetheless, economic analysis results are, on the surface, more inter-relatable than SVR assessment results. SVR assessments are generally case specific, in that the findings of a given assessment only apply to the region targeted by the study (Schmidtlein, Deutsch, Piegorsch, & Cutter, 2008). Furthermore, the assessment units are often ranked and categorised into ‘high’ and ‘low’ risk, but these are only relevant within the study area, and do not reflect an absolute measurement of risk. However, they provide a more detailed representation of the social conditions of the unit being assessed, and can include indicators which may be more important to regional managers than the economic value alone.

Brander et al. (2014) describe OA as an economic externality of CO_2 emissions which, along with climate change, is not currently reflected in the costs of fossil fuel combustion. These authors, along with many others (e.g., Talberth & Niemi, 2017), further suggest that ideal pricing of CO_2 emissions should reflect these real externalities in order for markets to respond accordingly. Several of the economic analyses had the specific goal of estimating the costs of OA in order to contribute to determining the total costs of current anthropogenic CO_2 emissions (L. M. Brander et al., 2014; Narita & Rehdanz, 2016; Narita et al., 2012). Narita & Rehdanz (2016) suggest that vulnerability(/risk) assessments should have an application for short term management and adaptation planning. But due to the lack of specific value attributed to losses, they go on to suggest that risk assessments are not directly applicable to discussions on global climate change mitigation strategies.

This perspective seems dependent on a worldview that assumes economic theory and market-based solutions will be able to resolve all aspects of climate change-related impacts on society. However, even if all climate change impacts (including OA) were accurately and completely priced, the impacts will persist and will be experienced unevenly around the world – though on a global scale, collective benefits of carbon emissions would be exactly equal to the collective costs. In this context, results of SVR assessments are particularly valuable as they can help highlight geographies and societies that will bear the unequal burden of OA and other climate change impacts. SVR assessments can also identify where mitigation actions should be prioritised (whether carbon is priced or not) as well as indicate factors of risk or vulnerability which can be acted upon (Cardona et al., 2012; Hajkowicz, 2006).

Relative rankings offered by the SVR assessments do not rely on making explicit predictions, as required in economic assessments, and therefore, can be informative in situations where specific economic valuation is difficult (L. M. Brander et al., 2014). Narita & Rehdanz (2016) also acknowledge that an advantage of risk assessments lies in the fact that they do not require making estimates based on economic projections, and suggest that they can rely more directly on biophysical science to drive their conclusions. In contrast, economic assessments must also rely on models of future economies which adds an additional layer of uncertainty.

Economic analyses present a single aspect of society that may be impacted by changes in fisheries access. While revenue is likely the main motivation for participation in a fishery, a change in the value of the fishery does not reveal the wider social importance. In some fisheries, harvest and production is highly concentrated into a small number of large enterprises; while other fisheries are distributed among many smaller operators and may support many more households and communities. In an assessment of the change in value of a fishery there would be no differentiation between these two scenarios. SVR assessment approaches, while not directly inter-comparable between studies, can specifically account for critical differences in the industry and social make-up of communities and account for additional factors which may increase or decrease the apparent impact from a change in fisheries. Both assessment approaches have applicability in different scenarios. Economic analyses can be used to more explicitly measure the total cost of carbon and help to drive market-based action to reduce total carbon emissions (Talberth & Niemi, 2017). However, in making management decisions and responding to where impacts are likely to be most severe, SVR assessments can be instrumental.

2.5.2 Future Research and Next Steps

Falkenburg and Tubb (2017) proposed that finer-scale economic studies could be nested within global assessments. Brander et al. (2014) also suggested that global-scale analyses can identify nations or oceanographic regions that are poised to be most affected by OA, but that local/regional scale assessments will be important for identifying specific areas most at risk. Assessments at different scales could also be used in sequence to fine tune where mitigation and adaptation resources are best allocated. This concept could be expanded to synergise between the economic and SVR assessment methodologies. To this end, a global-scale study such as that undertaken by Cooley et al. (2012) could be used to rank nations in terms of their vulnerability to OA, and those found to be at greater risk/more vulnerable could then be more closely examined with methods that place a greater focus on the most relevant social components for a given community. In parallel, nations that are found to be less vulnerable could still be assessed at finer scales through methods that present more contextually meaningful results, such as Moore's (2015) assessment of potential costs to consumers. A combination of nested methods could ultimately present findings at a high resolution, with results that can be meaningful to managers and policy-makers at federal, provincial/state and/or municipal levels.

To more fully address the human impacts that may result from OA, effects beyond fisheries will also need to be addressed. This review focused on fishery-related impacts so that the methods used by the studies that were discussed were more directly comparable and had similar goals. However, there is a small amount of literature currently available for assessing the social impacts related to how OA may affect other ecosystem services and components (e.g., economic costs from OA-driven degradation of coral reefs (L. M. Brander et al., 2012)). Further research should consider alternate pathways for OA to affect human societies and expand this area of understanding.

Despite the growing body of literature on the socioeconomic implications of OA, there is still enormous scope for advancements. Regardless of the socioeconomic approach followed, all of the currently available studies noted that a major weakness lies in the large degree of uncertainty surrounding biological responses to OA. This uncertainty is compounded by the fact that OA will not occur independent from, but alongside, other aspects of climate change (e.g., temperature change) – currently understanding of how these stressors will interact is lacking. Furthermore, the majority of biological response studies only assess single species responses, as such, an understanding of cascading ecological impacts is also lacking from literature. Most of the studies caution that their results could be either conservative or exaggerated, due to the limited nature of the underlying OA data. In these instances, the preferred perspective was to follow a pre-cautionary approach and view the results as conservative with respect to total OA impact.

2.5.3 Socioeconomic Assessment of Ocean Acidification in Atlantic Canada

Atlantic Canada is a region where fisheries are dominated by shellfish production and there is a relatively high social dependence on fisheries activities compared to other provinces in the country (DFO, 2004; Statistics Canada, 2011). The region therefore presents an ideal case-study for investigating the socioeconomic impacts of OA. In this setting an economic analysis would be less effective than an SVR assessment because the provinces in the region have significantly different populations and economies, as well as varying dependences on fisheries. Therefore, a change in the value of a fishery in one province would be potentially more (or less) impactful than an equal change in another province. The broader social context addressed by SVR assessment frameworks allows for more relevant socioeconomic indicators to identify which provinces are more threatened by change, rather than relying solely on a projected change in revenue or value from a given fishery. Therefore, the main goal of this thesis is to assess the risk posed to Atlantic Canada using a risk assessment framework, linked with a biophysical model, to investigate where the social impacts that result from OA and climate change driven fluctuations in future shellfish landings are likely to be most severe.

Chapter 3 Socioeconomic Risk from Ocean Acidification and Climate Change Impacts on Atlantic Canadian Fisheries

3.1 Introduction

Ocean acidification is a facet of climate change driven by increasing CO_2 concentrations in the atmosphere and ocean (Doney et al., 2009; Orr et al., 2005). Only relatively recently was it widely recognised as a potential biological stressor developing in the marine environment. Since then, research addressing its biological implications has expanded rapidly (Kroeker et al., 2013, 2010; Orr et al., 2005). Early findings have been wide-ranging and variable, with some species showing positive physiological responses and others responding negatively. Moreover, responses have also been seen to vary within species (Kroeker et al., 2010; Ries et al., 2009). Despite the sometimes contradictory insights from recent research regarding potential impacts of OA, it is widely agreed that invertebrate species that rely on calcium carbonate shells and exoskeletons will likely be the first to express negative responses partly because negative impacts have already been observed during upwelling events in the Pacific Northwest of the United States and Canada (Doney et al., 2009; Gazeau et al., 2007; Kroeker et al., 2013; Orr et al., 2005; Waldbusser et al., 2014; Washington State, 2012).

Seafood is a critical dietary component for billions of people and is a highly traded commodity (FAO, 2016; Smith et al., 2010), and changes in production could therefore have significant impacts for many and diverse communities. There is a body of emerging literature seeking to address how OA may affect these relationships by linking projected biological responses to OA with human systems. To date, socioeconomic OA impact research has been undertaken at a range of scales and resolutions from international (Cooley et al., 2012; Narita et al., 2012) to national (Ekstrom et al., 2015; Heinrich & Krause, 2016; Moore, 2011) and sub-national (Mathis et al., 2015).

Canadian seafood production is concentrated in Atlantic Canada, with over 80% of total landings (Figure 3) and over 85% of the commercial fishing fleet based in the region (DFO, 2017c). Within Atlantic Canada, shellfish have become an increasingly important component of capture fisheries, accounting for nearly 50% of regional landings by weight and over 75% of total landed value (DFO, 2017f; Divovich et al., 2015) (Figure 8). At the same time, Atlantic Canadian provinces and communities represent some of the least wealthy segments of the Canadian economy, with most of the provinces being net recipients of federal equalization payments as well as receiving overall above average per capita federal support (DFC, 2017). Furthermore, most of the provinces in the region have comparatively rural populations, with many relatively small communities which are more highly dependent on employment from natural resource-linked sectors such as fisheries (DFO, 2004; Statistics Canada, 2011). Given the high contribution of shellfish to total seafood production, together with the socioeconomic background of the provinces in the region, Atlantic Canada presents a highly relevant setting for investigating how potential OA-driven changes in fisheries might affect human communities. This chapter addresses this situation, and uses a biophysical model linked with a risk assessment framework to investigate how OA and climate change might drive shifts in future availability of fisheries resources for Atlantic Canada, and which provinces in the region are most likely to be affected by these shifts.

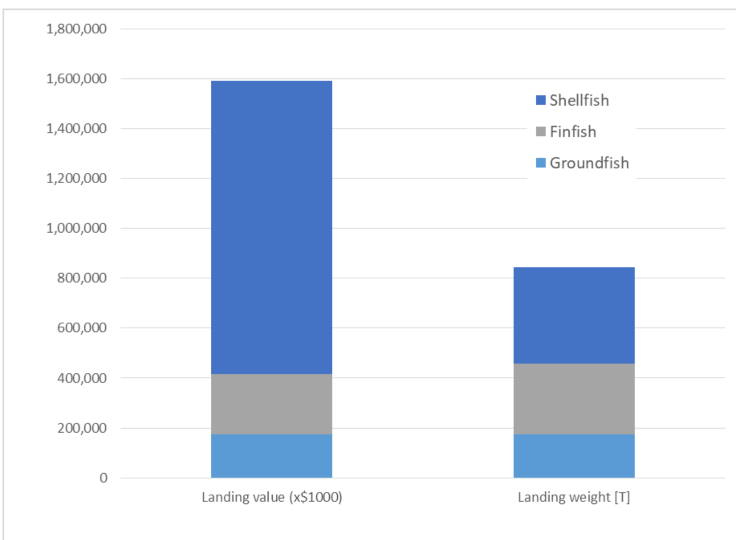

Figure 8. Average annual landing weights and dollar values by species group for Atlantic Canadian fisheries for 1991-2010. Dollar values are normalised to year 2000 dollars and are in thousands of dollars (both data types use the same axis). Includes production from aquaculture. Note DFO defines ‘finfish’ as pelagic vertebrate fishes, and ‘groundfish’ as demersal vertebrate fishes. ‘Shellfish’ effectively encompasses all marine invertebrate species. See Appendix 1 for provincial landing weights and values.

3.2 Methods

3.2.1 Scope

3.2.1.1 Study Area

'Atlantic Canada' typically refers to four Canadian provinces: New Brunswick (NB), Newfoundland and Labrador (NL), Nova Scotia (NS) and Prince Edward Island (PEI). The province of Quebec (Que) also borders the Atlantic Ocean but it is often not included as part of 'Atlantic Canada' due to significant demographic and cultural differences. However, for the purposes of this research, 'Atlantic Canada' will also include Quebec, as the province operates fisheries in Atlantic waters and is therefore exposed to potential changes in fisheries (Figure 9).

All Canadian marine waters fall under the jurisdiction of Fisheries and Oceans Canada (DFO). In Atlantic Canada, marine waters are divided into four management areas: Newfoundland and Labrador, Maritime, Gulf, and Quebec (Figure 9). In this analysis the Gulf and the Quebec management areas are treated as a single unit (henceforth, collectively referred to as the Gulf management area). This is because both management areas are comparatively small and the DBEM used to project distribution changes does not have the resolution to differentiate between these areas. The provinces of NS and NB border two management areas (Maritime and Gulf; Figure 9), and report landings for each area separately.

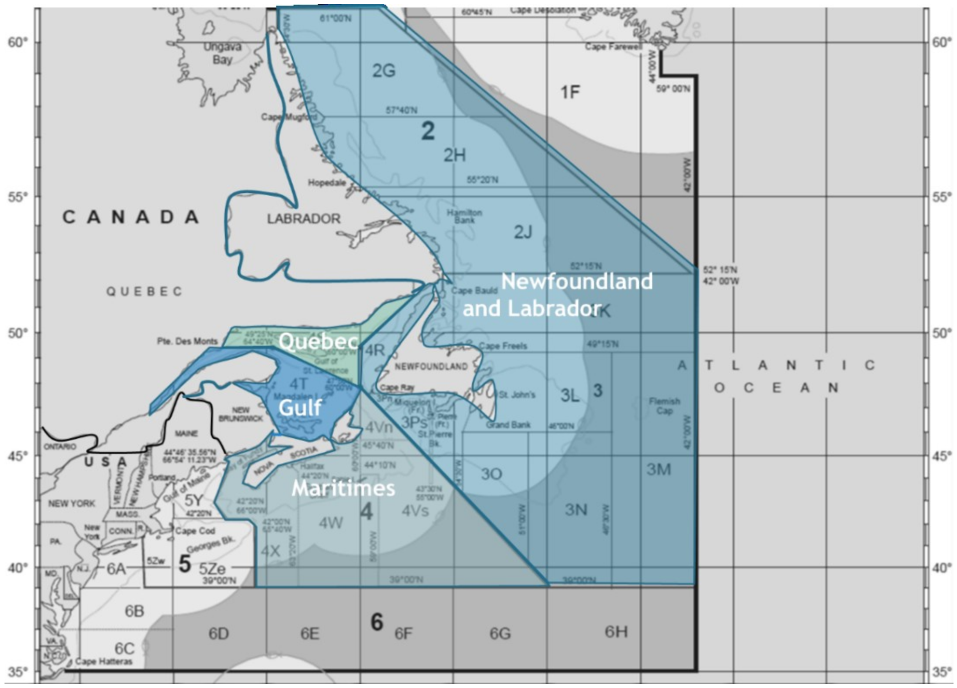

Figure 9. Eastern Canada as it relates to the Northwest Atlantic Fisheries Organization (NAFO) with DFO management areas overlaid. For clarity provincial borders have been emphasised in dark blue, U.S. and Canada border has been emphasised in black. In this assessment Gulf and Quebec management areas were treated as a single unit (Figure modified from (NAFO, n.d.))

3.2.1.2 Fisheries Data

A subset of commercially harvested species was selected to assess the potential OA impacts on fisheries and, ultimately, Atlantic Canadian communities. The selection of species was based on a combination of: a) their current commercial importance (including capture fisheries and aquaculture production) in Atlantic Canada; and b) the current understanding that shellfish species are more susceptible to OA than finfish species (Kroeker et al., 2013, 2010).

To guide the selection of the specific fisheries that would be the focus of this research total annual landings for all fisheries in each province and management area were compiled. Fisheries landings and values data were collected from DFO statistics (DFO, 2017f). Annual landing weights from 1991-2010 were averaged to a obtain baseline annual weight for the year 2000. Landing values for each year were normalised to year 2000 dollars using the consumer price index from the Bank of Canada (Bank of Canada, 2017), and similarly averaged across 1991-2010.

The compiled landing values included aquaculture production. While the DBEM projections are not directly applicable to aquaculture, the production values were included here because: a) the majority of shellfish aquaculture in the region relies on wild populations for recruitment (Isabelle Tremblay, personal communication, October 31, 2017). Consequently changes that affect wild population distributions will also likely affect current levels of aquaculture production. And b) DFO data for American oyster merges aquaculture and wild harvest production values; and PEI blue mussel data reporting changed from being reported as capture to being reported as aquaculture midway through the baseline time period. Aquaculture production data is available at a provincial level but does not differentiate between management areas. Therefore, the data for NS shellfish aquaculture production values were evenly divided between the Gulf and Maritime management areas because aquaculture production in NS was distributed across the whole province. For NB, shellfish aquaculture was all counted under the Gulf management area because NB shellfish aquaculture is concentrated there (in contrast to NB finfish aquaculture, which occurs in the Bay of Fundy (i.e., the Maritime management area)).

3.2.2 Biophysical Impact Modeling

3.2.2.1 Model Selection

To highlight potential impacts driven by OA on the fishery sector, shellfish fisheries were the focus of this research as current evidence suggests that these species are likely to respond more consistently and rapidly to the effects of OA (Kroeker et al. 2010, Kroeker et al. 2013). However, the wider context, in which climate change-driven shifts in species abundances and distributions occur, cannot be overlooked. The interactions between OA and other climate factors are still not well defined, although, literature on the subject is expanding (Crain, Kroeker, & Halpern, 2008; Ghedini & Connell, 2017; Kroeker et al., 2017). It was nonetheless decided that inclusion of the climate change related impacts along with OA would present a more representative estimation of the potential future of Atlantic Canadian fisheries. A DBEM (Cheung et al., 2011; Cheung, Lam, et al., 2008), which predicts future species distributions based on habitat suitability and life-history, was selected as the model from which the potential changes in landings in Atlantic Canada would be derived.

3.2.2.2 Model Data Processing

Data outputs from the DBEM were made available by William Cheung and the Changing Ocean Research Unit at UBC. The DBEM interprets habitat preferences for species based on current (and past) distributions, and then uses climate and oceanographic models to predict where suitable habitat will occur in the future; these are combined with population dynamic models to predict future distributions and abundances of marine species. For a summary of the model see Chapter 1, section 1.4. The model outputs used here incorporated OA as an impact on growth and survival for mollusc and crustacean species. Specifically, mollusc and crustacean species groups had different impact levels per unit change in pH based on the mean impact findings of Kroeker et al. (2013, 2010) (Cheung et al., 2011; Lam et al., 2014; Tai et al., In prep.).

Outputs from the model were provided as annual species-specific catch potentials (a proxy for maximum sustainable yield) for cells on a global half-degree latitude by half-degree longitude grid for the period of 1950 to 2100. For each species and year combination, there were 12 datasets generated by the model representing two RCP climate scenarios (RCP 2.6 – 'highly mitigated CO_2 emissions' and RCP 8.5 – 'business as usual CO_2 emissions'); two OA treatments ('with OA'; and 'without OA'); and using three separate climate models (NOAA's Geophysical Fluid Dynamics Laboratory (GFDL-ESM), Institute Pierre Simon Laplace Climate Modelling Centre (IPSL-ESM) and Max Planck Institute for Meteorology (MPI-ESM)) (Table 4).

Table 4. Schematic representation of data configuration for each species, *s*. Each of the 12 configurations yielded a separate global distribution of catch potential in half-degree latitude by half-degree longitude cells for each year, *Y* that was modelled (1950-2100).

<table>
<tr><td rowspan="12">Species s, Year Y</td><td rowspan="6">RCP 2.6</td><td rowspan="3">with OA</td><td>GFDL</td></tr>
<tr><td>IPSL</td></tr>
<tr><td>MPI</td></tr>
<tr><td rowspan="3">without OA</td><td>GFDL</td></tr>
<tr><td>IPSL</td></tr>
<tr><td>MPI</td></tr>
<tr><td rowspan="6">RCP 8.5</td><td rowspan="3">with OA</td><td>GFDL</td></tr>
<tr><td>IPSL</td></tr>
<tr><td>MPI</td></tr>
<tr><td rowspan="3">without OA</td><td>GFDL</td></tr>
<tr><td>IPSL</td></tr>
<tr><td>MPI</td></tr>
</table>

Data extraction for the species of interest along with further data manipulation was performed using Mathworks MATLAB, version R2015b, and Microsoft Excel 2013. The 12 global datasets for the selected species were first truncated geographically to include only data from grid cells corresponding to Canada's Atlantic EEZ (NAFO areas 2, 3 and 4; Figure 9). Modifying methods from Cheung et al. (2010), 20 year running means were calculated for each decade (i.e., for year 2000, values were averaged from 1991 – 2010) for each species' 12 geographically truncated datasets in order to smooth interannual variability from climate model projections. This was performed for each grid cell in the truncated datasets.

To limit variability resulting from individual climate model projections for each of the species-RCP-OA treatments (i.e., first three columns of Table 4) the median of the three climate model (i.e., column 4 of Table 4) values for each grid cell was selected to create a median dataset for further analysis. The median of the three climate models was selected in favour of the mean, because median values tended to be more moderated than mean values, but both were similar[7]. These processes of data smoothing and condensation resulted in a total of four datasets (two RCP scenarios and two OA treatments) for each of the selected species. The catch potentials were then summed across the DFO management areas (as per Figure 10) for each decade. Minor exceptions to this process were necessary where the median dataset projected future abundance of a given species in a management area dropping to zero. In these instances, the mean of the management area aggregated data from the three climate model values was substituted.

[7] The following steps were also performed for the datasets generated from the individual climate models (these are available in Appendix 2 – Table A2.2)

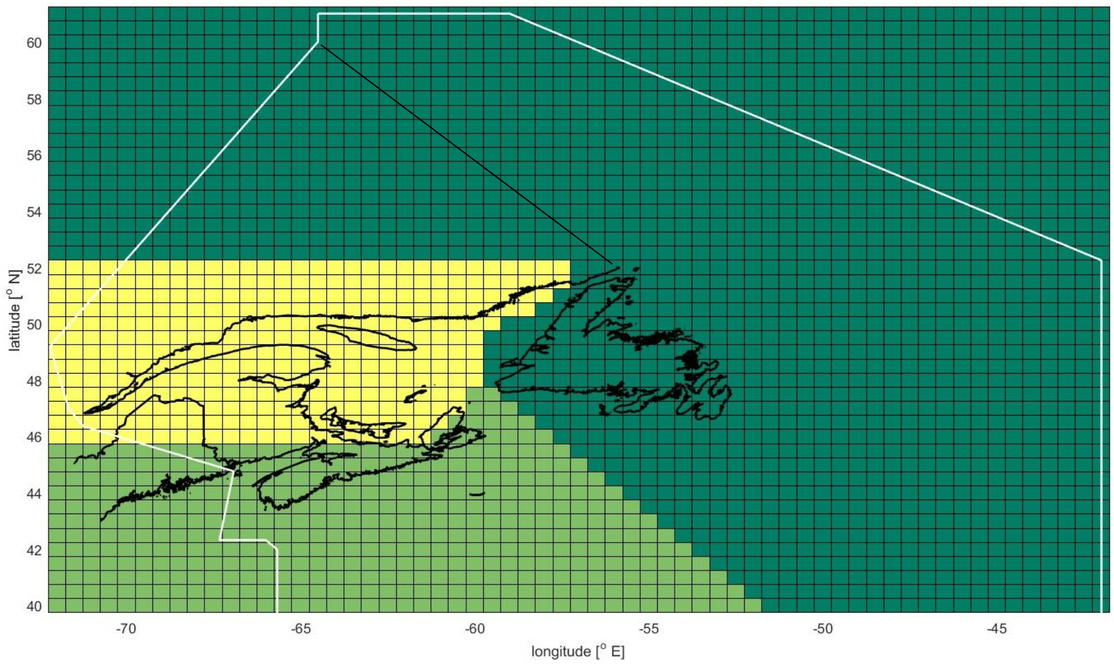

Figure 10. Geographic coverage of datasets. Grid lines indicate individual half-degree latitude/longitude data cells. White boundary line outlines study area (i.e., Canadian Atlantic EEZ). Data outside of this boundary was truncated from the datasets before aggregating management area data. Cell colours define data aggregation layer for DBEM outputs: Newfoundland and Labrador management area is represented by dark green cells; Maritime management area is represented by light green cells, and Gulf (and Quebec) management areas are represented by the yellow-green cells. Inclusion of land area within data aggregation is irrelevant, as these cells do not contain any data. Coastline data from: (M_Map, 2014).

As the DBEM data represents changes in catch potential, the absolute values of model outputs were not interpreted directly as future catch values. To estimate future landings, DFO reported landings (as per section 3.2.1.2) were multiplied by the relative change indicated by the outputs of the DBEM. The year 2000 was used as the reference year against which future changes in potential landings were assessed. Relative change in modelled catch potential for each species in subsequent decades was calculated for each DFO management area, as well as for the entire Atlantic Canadian region using Equation 3. To assess patterns across the region, the relative change for each grid cell was also calculated.

Equation 3

$$\Delta_{A,Y,r,s} = \frac{Y_{data} - 2000_{data}}{2000_{data}}$$

Where $\Delta_{A,Y,s}$ is the relative change compared to the year 2000 for management area A, in decade Y, under RCP scenario r, and species s.

3.2.2.3 Analysis of Potential Future Landings

Two future time-steps, representing the middle and end of the 21st century (2050 and 2090, respectively) were selected as endpoints for the assessment of changes in landings. Values for both time-steps were calculated relative to the reference year (2000) (i.e., the relative change for 2090 is relative to the year 2000 data, not the 2050 data).

The coupled impact of climate change plus OA was determined to be a more relevant focus for the investigation of potential impacts across Atlantic Canada. Therefore the treatment with OA was used for the main analysis. The 2050 and 2090 potential landings for each species within each management area were compared with current landings. Differences in future landings under the two climate scenarios (RCP 2.6 and RCP 8.5) were also compared to identify how different emission scenarios might affect future fisheries. Individual species changes, as well as cumulative changes across species, were investigated within and between the management areas.

3.2.3 Constructing the Socioeconomic Risk Index

A risk assessment framework was developed to evaluate the socioeconomic risk posed to Atlantic Canada by changing fisheries landings driven by ocean acidification and climate change. A range of approaches to modelling potential future risk posed by hazards such as OA and climate change have emerged in the literature in recent years (Cardona et al., 2012; Ekstrom et al., 2015; Mathis et al., 2015). The framework constructed here broadly follows the methods of Mathis et al. (2015), wherein risk was defined as the intersection of the hazard, exposure to the hazard, and vulnerability to the hazard. Vulnerability was itself composed of two sub-components: sensitivity and adaptive capacity (for a summary of risk assessment theory and terminology see Chapter 1, Section 1.5).

In the framework developed for Atlantic Canada the exposure component represented the expected coupled OA and climate change-driven effect on the fisheries as determined through analysis of DBEM outputs. The sensitivity and adaptive capacity were used to describe the social factors that may bolster or obstruct communities' responses to changes in fisheries. In total, the risk index was informed by six separate indicators: one for exposure, two for sensitivity, and three for adaptive capacity (Figure 11). Within each individual component of the framework, the sub-components or indicators were equally weighted. This implies that indicators are equally important. This may not be the case, but without strong support for preferential weights, equal weighting was selected as the most straightforward and neutral system (Hajkowicz, 2006; OECD, 2008) (Figure 11).

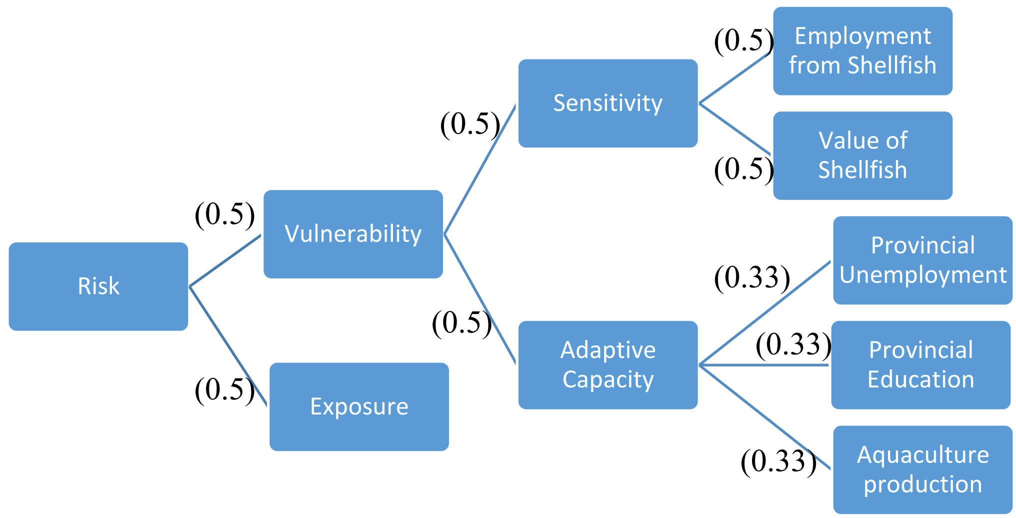

Figure 11. Framework for the risk index demonstrating relationship between components. Elements at each branch are equally weighted to other items at the same level as indicated by bracketed numbers.

Even though a cumulative risk index was constructed, the biophysical and social factors should be seen as representing two distinct aspects of risk (Cardona et al., 2012; Ekstrom et al., 2015). The combined index represents where these two dimensions overlap to have the highest impact. The biophysical indicator (i.e., exposure) represents where the most notable changes in the resource will occur; and the social indicators (i.e., vulnerability) highlight where changes are likely to have the highest impact.

Two of the provinces, NB and NS, border two separate DFO management areas (the Gulf and the Maritime management areas; Figure 9). For the purposes of the risk analysis, these two provinces were sub-divided into gulf (NB-gulf; NS-gulf) and maritime (NB-mar; NS-mar) provincial sub-sections. Both sub-sections used the same social data because these data were not available at a resolution that would have allowed separate treatments, but the sub-sections were exposed separately to potential changes in both relevant DFO management areas.

3.2.3.1 Exposure Indicator Construction

In this analysis, the most significant divergence from previous risk/vulnerability assessments of potential impacts of OA was the use of a biophysical model to inform the exposure and hazard components of the framework simultaneously. Other assessments have used an expected change in ocean chemistry to represent OA (the hazard) and used this in concert with the importance of OA susceptible fisheries (the exposure) to link the OA phenomenon with fisheries (Cooley et al., 2012; Ekstrom et al., 2015; Heinrich & Krause, 2016; Mathis et al., 2015). In this assessment, the biophysical model directly applied the expected climate change impact (including OA) onto the relevant fisheries. This allowed the change in fisheries landings to be explicitly incorporated into the framework. Thus the hazard and exposure terms were essentially incorporated into a single term (this combined component is referred to as 'exposure' throughout this assessment).

A single indicator to account for all of the assessed species was designed. To reflect the very different current economic contributions of individual species to total landed value, the proportion of the total shellfish value from each species was calculated and multiplied by its DBEM-predicted relative change for each management area. These value-scaled changes were summed across all species (Equation 4). This was repeated separately for 2050 and 2090 and both RCP climate scenarios to obtain four exposure indicator scores per management area. From here forward, the term 'quartet' will be used to refer to the four exposure scenarios, as well as the subsequent risk scores.

Equation 4

$$E_{A,Y,r} = \sum \left[\Delta_{A,Y,r,s} \times \left(\frac{V_{A,s}}{V_{A,tot}} \right) \right]_s$$

Where $E_{A,Y,r}$ is the exposure indicator score for area A, in time-step Y, and RCP scenario r. $\Delta_{A,Y,s}$ is the DBEM predicted relative change in landings for time-step Y, RCP scenarios r, and species s (from Equation 3). $V_{A,s}$ is the year 2000 average annual value for species s, and $V_{A,tot}$ is the total annual landings of the seven species value for the management area (Table 5).

The risk index was constructed so that high scores imply higher risk. To align the exposure scores with this, the values were multiplied by (-1) so that losses in potential future landings became positive. Additionally, in order to combine the separate indicators, it was necessary to first normalise each indicator to a score between 0 and 1 (Hajkowicz, 2006; OECD, 2008). The 12 exposure values (three management areas, each with a quartet of scores) were therefore normalised to score between 0 and 1, so that the most extreme decline scored 1, and the most extreme gain scored 0. In the risk index the exposure scores for management areas were combined with the social vulnerability of adjacent provinces – therefore the Gulf and the Maritime management areas' scores were applied to more than one province (NB-gulf, NS-gulf, PEI, and Que; and NB-mar, NS-mar; respectively) while the Newfoundland and Labrador management area scores were only applied to NL.

3.2.3.2 Social Vulnerability

Vulnerability represents a province's reliance on fisheries (sensitivity) as well as the community's broader ability to respond to, and absorb, changes (adaptive capacity). Vulnerability was assessed at the provincial level as this was the finest level of political organization for which most relevant data were consistently available. Additionally, to align the social data with the exposure data, where possible the year 2000 was used as the baseline. However, some of the data (e.g., crew size) was drawn from reports which are not published annually, consequently these data are from slightly earlier/later years.

3.2.3.2.1 Sensitivity

Sensitivity of each province was based on the importance of the assessed fisheries to each province's economy and social structure. Each province was scored on two indicators: a) the value derived from the seven species relative to total provincial GDP, and b) the employment directly associated with harvesting the selected species (Figure 11). The first factor was assessed using year 2000 total annual landed value of the assessed species as a fraction of total provincial GDP (averaged between 1997 and 2003, and all dollar values were normalised to year 2000 dollars (Statistics Canada, 2017c)). Scaling the fisheries value relative to provincial GDP yielded indicator scores that were more relatable between provinces, which have a broad range of economic scales.

It is difficult to obtain reliable official employment estimates for individual fisheries because of their seasonality. This type of data is further convoluted by different methods of reporting employment statistics. For example, these are often presented as full-time equivalent jobs (a measure of the number of hours of work) in government reports. This is not a particularly relevant metric for seasonal employment – especially when it can generate enough income that additional employment is not necessary to support a desired quality of life (as can be the case for high-value fisheries). This study attempted to reflect the number of people employed with fisheries as their main source of income.

An estimate of employment was derived from the number of licences per species multiplied by average crew sizes for the applicable fishing fleet. Licence numbers were obtained from DFO statistics (DFO, 2017b). Where possible crew size estimates were collected from industry reports and assessments [8,9] (L. Brander & Burke, 1994; DFO, 2007; Gardner Pinfold Consulting, 2006a, 2006b; Stevens et al., n.d.) as well as through discussion with Michael Gardner, of Gardner Pinfold Consulting (personal communication, September 11, 2017). Most crew size estimates were from studies conducted in the early 2000s and did not necessarily align with the 2000 baseline year. This data misalignment is unlikely to affect overall results as average crew size within a fishery should remain relatively stable through time – at least while value of the fisheries remains relatively stable (Michael Gardner, personal communication, September 11, 2017). Where crew size estimates were not available for a given species within a specific management area, the average crew size from fisheries for that species from other areas was used. Total employment in the relevant fisheries was summed for each province and divided by provincial population.

[8] Several fisheries in Atlantic Canada are distributed between "inshore" and "offshore" fisheries, with the main operational difference being vessel size. For most fisheries employment is concentrated in the inshore fishery, while a few larger vessels (with correspondingly larger crews) operate in the offshore. This was most relevant for the sea scallop (*Placopecten magellanicus*) fishery in NS. To account for this the NS data included estimates from two reports for offshore licence numbers and crew sizes (L. Brander & Burke, 1994; Stevens et al., n.d.). The licence numbers were subtracted from the DFO reported number of licences (which do not differentiate between inshore and offshore) and calculated employment for both fisheries was summed.

[9] Stimpsons' surf clam (*Mactromeris polynyma)* licence data was collected from reports (rather DFO licences statistics) because DFO reported licence numbers represent all species of harvested clams, while the bulk of commercial surf clam harvest is concentrated under a very limited number of licences (DFO, 2002).

Aquaculture employment was not considered in this indicator because available data does not readily differentiate between finfish and shellfish aquaculture employment. Furthermore, employment from aquaculture is relatively minor compared to wild harvest employment. As a result employment dependence was set to zero for species whose production was derived entirely or almost entirely from aquaculture. For similar reasons estimates of secondary employment such as processing and retail were not considered in this assessment.

In many fisheries in the Atlantic region licences are not fully utilised. However, given the high value of the species being included in this study, it was considered reasonable to assume all or a very high proportion of all licences in these fisheries were active. However, crossover between fisheries, where a licence holder (as well as crew) may operate in more than one of the assessed fisheries was not accounted for (i.e., double counting was a possibility). Without a much more thorough social investigation this indicator provides an approximation of the employment derived from shellfish harvesting in the region. Assuming employment patterns in the industry are similar between provinces, using primary harvesting as a proxy for total employment should provide a reasonable first-order estimate. Furthermore, these values were used relative to each other and were not interpreted as representing absolute values of employment.

The scores obtained for both sensitivity indicators were normalised to score between 0 and 1, with the province where the fisheries are proportionally most important scoring 1 for each indicator. For each province the two indicator scores were averaged (Figure 11) to obtain a final sensitivity score with a maximum possible score of 1. A high score indicated a higher reliance on the fisheries, and a more sensitive social unit.

3.2.3.2.2 Adaptive capacity

To estimate the ability of each province to potentially respond to changes in fisheries landings, three indicators of adaptive capacity were compiled. Adaptive capacity is typically comprised of positive aspects of a society (Cardona et al., 2012), so the indicators were collected with higher scores indicating greater adaptive capacity.

In other socioeconomic assessments of potential OA effects on fisheries, an indicator representing alternative employment options has been used as a key component of adaptive capacity (Ekstrom et al., 2015; Mathis et al., 2015). As previously mentioned, due to the seasonal nature of fisheries employment in Atlantic Canada it is difficult to distinguish patterns of employment in fisheries as separate from employment in other sectors. Therefore, provincial unemployment rates were used to indicate the potential for alternative employment options to fishing (Heinrich & Krause, 2016; Mathis et al., 2015). Provincial rates were scaled against the national unemployment rate. Data were averaged across a 10 year period bracketing the year 2000 baseline for the assessment (i.e. 1996-2005 - (Statistics Canada, 2017a)). The reciprocal of unemployment values were used so that lower unemployment increased adaptive capacity.

Education within a society provides an indication of how well a community will be able to respond to changing conditions and has been used in other socioeconomic assessments of OA (e.g., Heinrich & Krause, 2016; Mathis et al., 2015). Higher education levels are generally seen as presenting more opportunities to adapt, while lower education rates limit options and capabilities to respond to change. The percent of adult populations with at least some post-secondary education[10] was used to represent the education level of the five assessed provinces (Statistics Canada, 2017b). As with the unemployment data, provincial education values were also scaled against the national values and were averaged from 1996 to 2005.

The final element that was considered as part of each province's adaptive capacity was the extent to which potentially OA-impacted species are currently farmed in the region. This is because aquaculture production is expected to be more resilient to OA and climate change than wild harvest, since many environmental conditions can be at least partially controlled or compensated for – especially when hatcheries are used to rear organisms through the most susceptible life-stages[11] (Clements & Chopin, 2016). Therefore a strong aquaculture sector could potentially strengthen a community's ability to respond to OA and climate change.

[10] Statistics Canada, education categories included were 'some postsecondary', 'postsecondary certificate', and 'university degree' (Statistics Canada, 2017b).

[11] Although this is currently not the case for most of Atlantic Canada it was seen as a potential pathway for adaptation in response to OA and climate change.

In Atlantic Canada there is no aquaculture production of crustacean species. Therefore, the aquaculture indicator was constructed to reflect only mollusc production[12]. First, the trend in mollusc aquaculture production was estimated for each province by comparing the average annual production for 2001 to 2010 against average annual production for 1991-2000[13]. Second, the fraction of total shellfish production (including both wild harvest and aquaculture) sourced from mollusc species (using the year 2000 average annual production) was calculated. These two terms were then multiplied together to obtain the indicator value (Equation 5).

Equation 5

$$A_p = \left(\frac{Aqua_{2001-2010}}{Aqua_{1991-2000}}\right) \times \left(\frac{Mol_{1991-2010}}{Tot_{1991-2010}}\right)$$

> Where A_p is the aquaculture indicator score for province *p*. *Aqua* denotes the average aquaculture production for the subscripted time-period; *Mol* indicates mollusc production; and *Tot* indicates total shellfish production.

As with the previous risk components, each of the adaptive capacity indicators were transformed linearly to score between 0 and 1. To align with the directionality of the other components in the framework the normalised scores were subtracted from 1 so that weak adaptive capacity indicators had higher scores (i.e., closer to 1) and contributed to risk. As with the indicators for sensitivity, the individual adaptive capacity indicators were treated as having equal weights. Therefore, the three adaptive capacity indicators (for employment, education and aquaculture production) were averaged to obtain a cumulative adaptive capacity score with a maximum potential value of 1.

3.2.3.2.3 Vulnerability

Sensitivity and adaptive capacity were treated as having equal weights, and their values were averaged to arrive at a social vulnerability score for each province, with a maximum potential value of 1 (Figure 11). Weighting the sensitivity and adaptive capacity components equal to each other indirectly applied different weights to their composite indicators (Figure 11). However, the concepts of sensitivity and adaptive capacity

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[12] In other geographies an aquaculture component of adaptive capacity would have to consider trends in species production and adjust the inclusion of an aquaculture indicator accordingly

[13] As previously noted, oyster production data does not differentiate between wild harvest and aquaculture production. Since most of its production is from aquaculture, all oyster production was included as aquaculture production for these calculations.

represent different aspects of social vulnerability and their overall contributions were treated as equally relevant.

3.2.3.3 Risk

To calculate the final risk index scores the exposure and vulnerability scores were equally weighted and combined. Each province (or sub-section thereof for NB and NS) received a quartet of risk scores following the quartet of exposure indicator scores that were applied. The 28 final risk index scores were divided into five categories of risk, and described as 'high,' 'moderate,' 'low,' 'minimal' and 'least' risk to OA and climate change. Each category represented six separate index scores, except the middle category (low), which only included four scores. Categories were not intended as absolute descriptors of the risk posed to the provinces, rather they describe relative levels of risk within the Atlantic Canadian region.

3.3 Results

3.3.1 Species

Shellfish make up nearly half of the total annual landing weight in Atlantic Canada. Furthermore, shellfish species typically receive a higher value per tonne than finfish, such that shellfish landings make up nearly three quarters of the total annual fisheries value (Figure 8). Shellfish landings are dominated by crustacean species, with northern shrimp (*Pandalus borealis)*, snow crab (*Chionoecetes opilio)* and American lobster (*Homarus americanus)* accounting for over 60% of the total shellfish landed weight (30%, 21%, and 12% by weight respectively; and 17%, 26% and 41% by landings value, Figure 12). However, some mollusc species also contribute substantially to landings, with sea scallop *(Placopecten magellanicus)*, Stimpson's surf clam (*Mactromeris polynyma)*, eastern blue mussel (*Mytilus edulis)* and American oyster (*Crassostrea virginica*) representing 19%, 7%, 5%, and 3% of the shellfish landing weight, respectively (9%, 3%, 2%, and 1% of total shellfish value, Figure 12). Remaining shellfish species that are harvested commercially combine to make up about 5% of the total shellfish landing weigh (and less than 2% by value). Due to their significant contribution to the total commercial fisheries production in Atlantic Canada, changes in landings for the above-identified species would represent a significant change for the value of the whole region's fisheries.

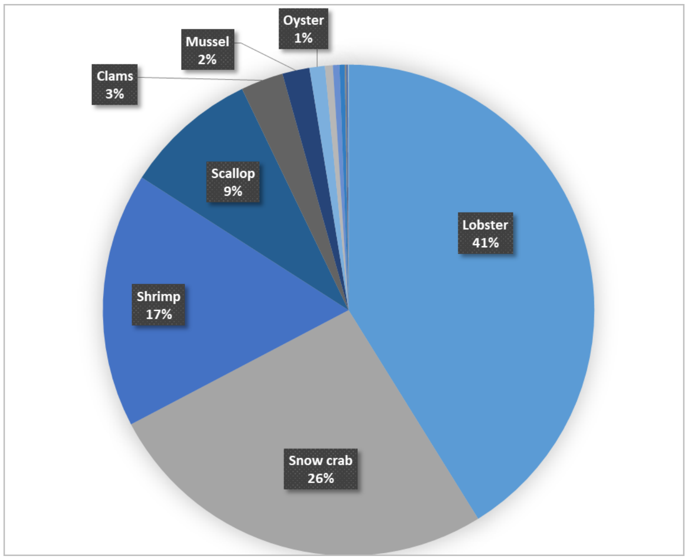

Figure 12. Breakdown of total shellfish landings value. Top seven most valuable species are indicated by data labels. The remaining shellfish species include other crab species (i.e., not snow crab), sea urchin, whelks and sea cucumber as well as 'other shellfish,' which together amount to 1.5% of the total value. (Appendix 1 includes the year 2000 (20 year average) landings values and weights by species and province).

3.3.2 Biophysical Results

3.3.2.1 Atlantic Canadian Regional Changes

Across the whole Atlantic Canadian region change in the cumulative net landings for 2090 are anticipated to be relatively neutral across the seven assessed species, regardless of the climate scenario (Figure 13). The largest relative changes are predicted for species with the lowest total landing weights in 2000: American oyster, eastern blue mussel and Stimpson's surf clam all have changes exceeding 15% (positive for oyster and mussel, negative for clams) under RCP 8.5, but all have region-wide landings under 30,000 tonnes (Figure 13). Thus, relatively large percent changes result in comparatively small absolute changes in landings for these species. In terms of absolute change at the regional scale, northern shrimp are projected to experience the largest absolute increases in production by the end of the century (+2,000T to +10,000T for RCP 2.6 and RCP 8.5, respectively, out to 2090). However, since northern shrimp represented the highest landing weight species in 2000, the relative gains are somewhat modest (2-8%, for RCP 2.6 and RCP 8.5, respectively). Conversely, snow crab had both high relative change and high absolute landings, with a predicted decline of 16-17% (RCP 2.6 and RCP 8.5, respectively) by 2090, on top of the second highest 2000 landed weight (~80 000 T) (Figure 13).

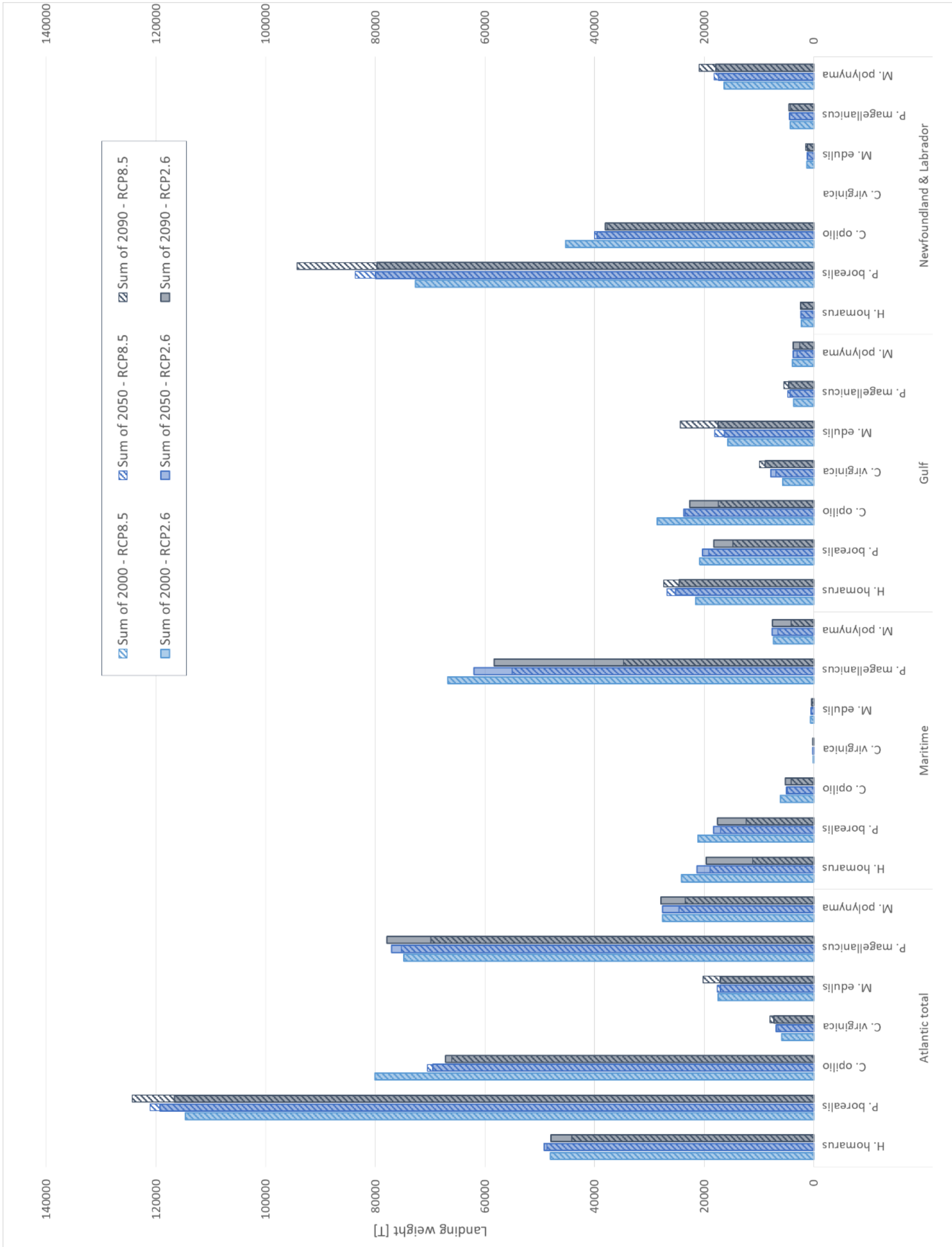

Figure 13. Changes in landing weight for projection scenarios for each species in the three DFO management areas and across the total Atlantic region. Each set of bars from left to right presents the year 2000 catch weight (pale blue), 2050 projected catch (dark blue), and 2090 projected catch (grey). Sections of the bar with diagonal hashes denote the RCP 8.5 climate scenario projections; the solid sections indicate the RCP 2.6 projections. The projected changes for the total regional landings are based on the relative change calculated across the whole study region and therefore do not present the sum of the projected landings from the individual management areas. DBEM relative change projections as percent change for each species in each time-step/climate scenario are available Appendix 2.

Results of the DBEM under the RCP 8.5 emissions scenario tend to show more extreme changes in potential future landings, whether positive or negative, than the predictions under the RCP 2.6 scenario (Figure 13). Similarly, the 2090 time-step tended to exhibit larger changes than the 2050 time-step (Figure 13). There were some instances where these trends were reversed (e.g., sea scallop across the entire region - (Figure 13)); however, they generally only occurred when projected impacts were very small. Relative changes that are very small, especially when the actual landing weights are also low, are best interpreted as no change or insignificant change. As a, small relative changes predicted by the model may be the result of interannual variability in the underlying climate models. Nevertheless, for some of the highly landed species (e.g., northern shrimp), even a small percent change implies a sizable change in tonnes landed.

When all seven species are considered in combination across the entire region out to 2090, total landings are projected to decline, if very slightly. Under the RCP 2.6 scenario a total loss of 6,400 tonnes is predicted relative to year 2000 landings (representing a 1.8% reduction in tonnage), and for the RCP 8.5 scenario the total projected loss nearly doubles to 12,200 tonnes relative to year 2000 landings (representing a 3.3% reduction in total landings) (Figure 13). In both scenarios this was mainly driven by the declines in snow crab, with the RCP 8.5 scenario results compounded by significant losses in sea scallop and Stimpson's surf clam (Figure 13).

While the cumulative changes across the region are minimal, on a sub-regional scale different patterns emerged. Latitudinal gradients appeared to be the main driver of species distribution with an overall northward trend apparent for most species (Figure 14). Overall, the Maritime management area was expected to see losses for most species (except Stimpsons' surf clam, which experienced minimal change in all scenarios). While in the Gulf and Newfoundland and Labrador management areas a mix of changes was anticipated, with overall increases slightly outweighing declines (Figure 13).

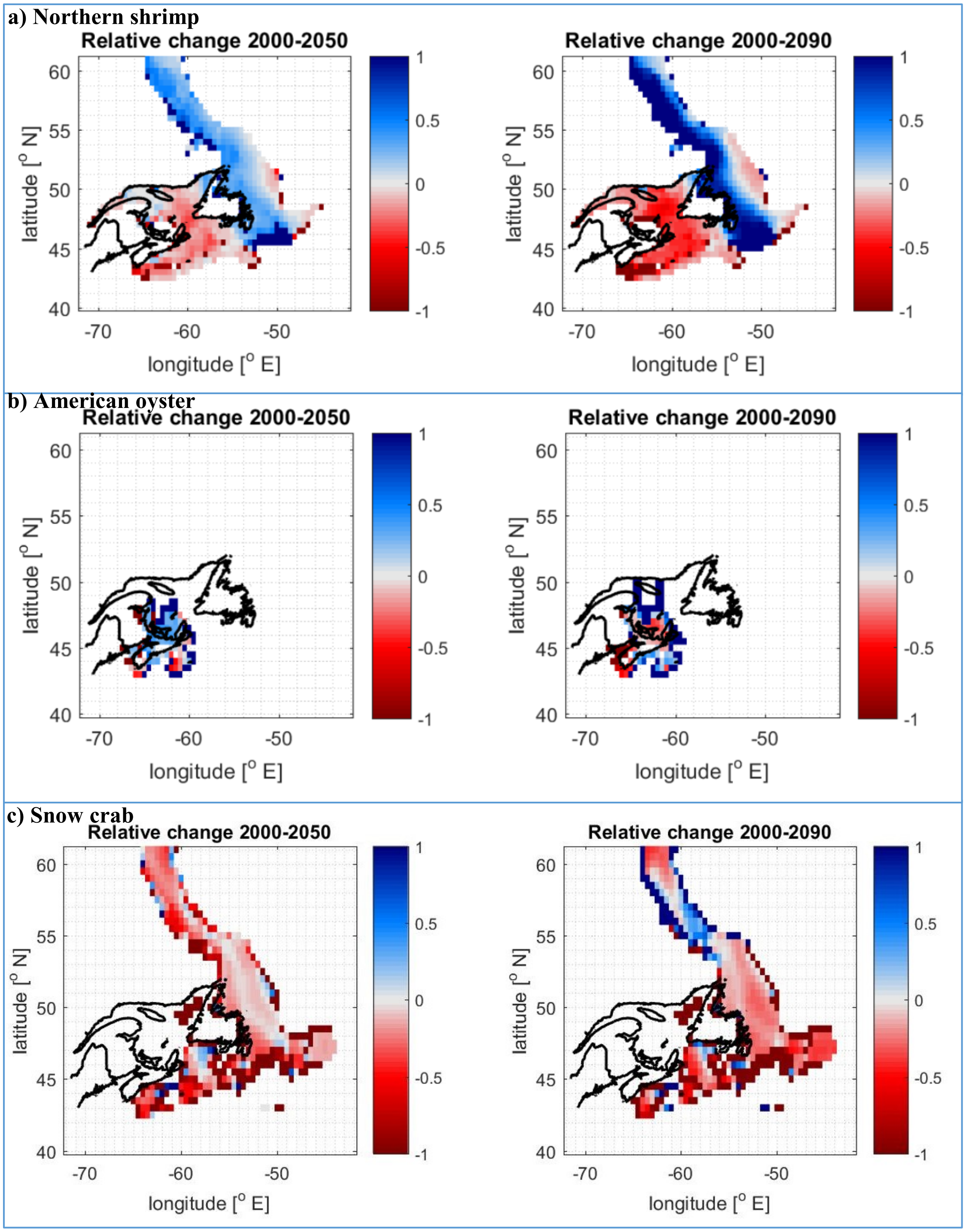

Figure 14. Relative changes in distribution for northern shrimp (a -top), American oyster (b - middle) and snow crab (c - bottom), for 2050 (left) and 2090 (right) under RCP 8.5, highlighting south to north trend of changing species distributions. Colour scales indicate relative change predicted by DBEM for each half-degree latitude by half-degree longitude cell and do not indicate absolute values. Darkest shades (in both directions) likely correspond to cells with low absolute values, where small absolute changes result in large relative changes (e.g., a shift of 1T to 2T results in a 100% increase, while a change of 10T to 15T only leads to a 50% increase). Changes that exceeded +100% were set to 100% to maintain coherent colour scales. See Appendix 3 for baseline DBEM distributions and figures for other species and climate treatments.

3.3.2.1.1 Modelled Ocean Acidification Impacts

When the difference between the model outputs ‘with OA’ were compared against the outputs ‘without OA’ the overall trends in landing changes largely remained the same. However, the OA effects in all scenarios for all species moderated increases and exacerbated declines (Figure 15). As expected given the implementation of OA in the model, mollusc species were more heavily influenced by the OA effect (eastern blue mussel are an exception where the OA impact was almost indistinguishable, possibly due to low overall abundance or the eastern blue mussel physiology in the model being more affected by other environmental factors). For the crustacean species the ‘with OA’ treatment tracked the ‘without OA’ treatment more closely, but were nonetheless expected to be somewhat lower throughout the century. Under the RCP 8.5 scenario the OA impacts were more pronounced than under the RCP 2.6 in all species (except eastern blue mussel) (Figure 15).

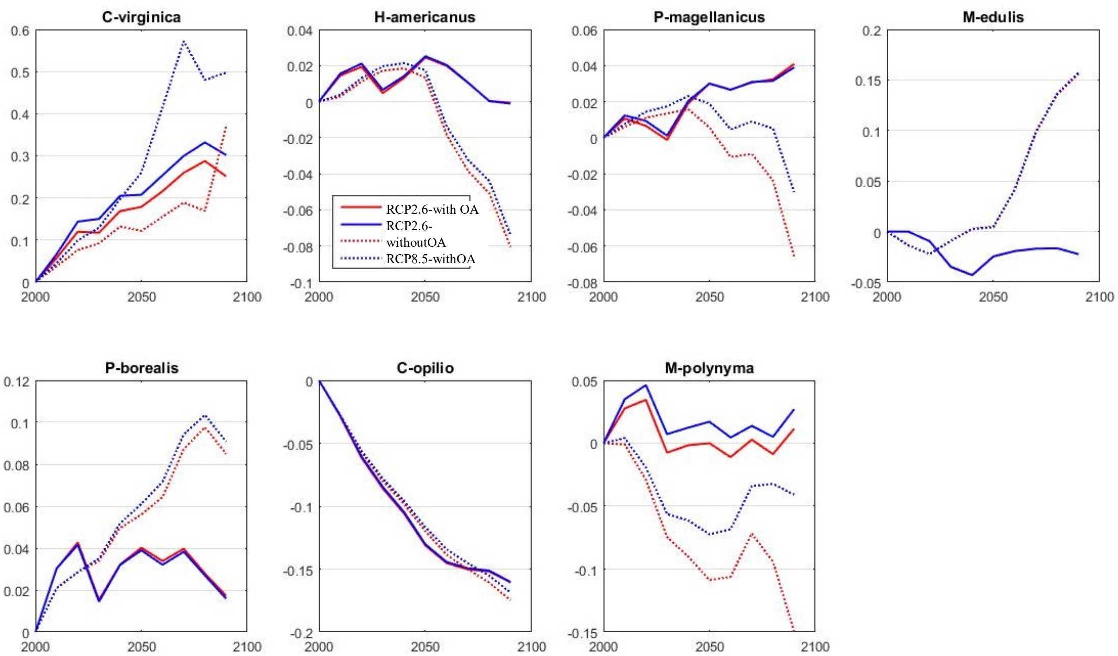

Figure 15. Comparison of model outputs with OA (red lines) and without OA (blue lines) through the century, under both climate scenarios (RCP 2.6 as solid lines, RCP 8.5 as dotted lines) for whole Atlantic Canadian EEZ. Values are relative changes in catch potential. Note different axes between sub-plots. (Appendix 3 contains relative changes in distribution for both ‘with OA’ and ‘without OA’ treatments)

3.3.2.2 Maritime Management Area

The three species in the Maritime management area with greatest landing weights in 2000 (American lobster, northern shrimp, and sea scallop) were all projected to decline under the quartet of exposure projections. Declines were more extreme in 2090 and under the RCP 8.5 climate scenario (Figure 13). American lobster is projected to experience the most significant decline in landings (54%). Slightly lower levels of decline were projected for sea scallop (48%) and northern shrimp (42%). Importantly, these projected declines for these commercially important species under the RCP 8.5 scenario are roughly double (or more than) the declines projected under the RCP 2.6 scenario (lobster 29%, scallop 13% and northern shrimp 17% in 2090 relative to 2000) (Figure 13).

The remaining species contributed relatively minorly to the Maritime management area landings. Under the RCP 2.6 scenario snow crab had the most extreme relative decline (21%) but the loss was somewhat smaller under RCP 8.5 (16%). Even so, this species contributed a comparatively small portion of the total landings to the Maritime management area in 2000 (Figure 13). Eastern blue mussel landings also decline under all treatments, but this species was nearly non-existent in the baseline data. The two species (American oyster and Stimpsons' surf clam) that had potential gains in landings also represent comparatively low total landings for the region. American oyster landings are essentially negligible with only ~140 tonnes of production in 2000, thus the ~10% increase predicted under RCP 8.5 for 2090 amounts to an approximate gain of 15 tonnes (Figure 13). Stimpsons' surf clam landings change negligibly under all treatments.

3.3.2.3 Gulf Management Area

Care should be taken when interpreting model projections for the Gulf management area, as its semi-enclosed nature with substantial freshwater input from the Saint Lawrence River means it is highly influenced by processes that are not well constrained by the climate models which inform the DBEM. Given the exploratory nature of this assessment as a starting point for anticipating future fisheries impacts in Atlantic Canada, the DBEM outputs were treated as a first approximation of potential future scenarios.

American lobster, eastern blue mussel, American oyster, and sea scallop are all projected to increase production moderately in the Gulf management area for the 2090 time step and under the RCP 8.5 scenario (Figure 13). Eastern blue mussel has the largest relative change, increasing 68% under RCP 8.5 by 2090 (12% under RCP 2.6) (Figure 13). Notably, the Gulf management area encompasses nearly all of the current mussel and oyster production in Atlantic Canada (mainly in the form of aquaculture). Therefore, changes in production for these species in this management area drive changes for the whole Atlantic Canadian region.

Snow crab landings in the Gulf management area appear to decline to zero very rapidly from nearly 30,000 tonnes in 2000 (representing ~35% of total snow crab landings in all of Atlantic Canada). This occurred because in two of the three climate models (the GFDL and IPSL) and hence the median dataset – the DBEM-modelled distribution in the Gulf in 2000 was minimal (Appendix 2 – Table A2.2 and Appendix 3 – Figure A3.1-u-x), and amounted to less than 1% of the total predicted landings for the whole Atlantic Canada region (Appendix 4). When the modelled projections of this small initial distribution fell to zero, it resulted in relative change of -100%. The outputs based on the third climate model (MPI) predicted higher initial abundances and therefore less extreme declines of snow crab in the area (reductions of 20% and 39%, for RCP 2.6 and RCP 8.5, respectively). Therefore, a mean of the three climate model impacts was adopted for the analysis of this species (Appendix 2 – Table A2.1). Under this approach snow crab is projected to decline in the Gulf management area by 17-18% in 2050 (RCP 2.6 and RCP 8.5, respectively) and 21-39% by 2090 (RCP 2.6 and RCP 8.5, respectively) from the 30,000 tonnes landed in 2000 (Figure 13).

3.3.2.4 Newfoundland and Labrador Management Area

The Newfoundland and Labrador management area not only encompasses the largest area of marine waters (Figure 8) but also has some of the highest landings, by weight, with over 50% of the total Atlantic Canadian catch for northern shrimp, snow crab and Stimpson's surf clam in 2000 (Figure 13). The patterns for northern shrimp and Stimpson's surf clams were similar for both scenarios and time-steps, albeit at different scales, as the shrimp landings were more than four times that of the clam. For both species under the RCP 2.6 scenario there were modest projected increases in landings by 2050 (10% and 6% for shrimp and clams, respectively). However, under this scenario projected production to the end of the century remained relatively unchanged, with shrimp staying at 10% above 2000 levels, and clams increasing slightly to 9% over 2000 landings. In contrast, under the RCP 8.5 scenario both species continued to increase in production through to 2090 (Figure 13).

As observed in the other management areas, snow crab was also projected to decline in the Newfoundland and Labrador management area. The declines were similar between the RCP scenarios, while the landings for 2090 declined slightly from the 2050 values, from 13-12% in 2050 (RCP 2.6 and RCP 8.5, respectively) to 16% in 2090 (for both climate scenarios).

The area was expected to see minor changes for lobster, scallop and mussel landings under the quartet of projections. Although, mussel production was an exception and was expected to increase 21% by 2090 under RCP 8.5. Nevertheless, all three species represent minor fractions of the region's landings and have much higher production values in the other two management areas.

3.3.3 Risk Assessment Results

3.3.3.1 Exposure

Scaling the DBEM-modelled changes in landings by species value in 2000 yielded an array of indicator scores representing the cumulative exposure from changes in the seven assessed species (Table 5). In each quartet of exposure scores, the 2090 time-step had higher exposure scores than the 2050 time-step (NL was an exception to this trend under the RCP 8.5 scenario) (Table 5). Since the RCP 8.5 climate scenario drove increased landings in some fisheries, the exposure scores corresponding to this climate scenario were higher for both the Gulf and the Newfoundland and Labrador management areas. However they were lower in the Maritime management area (and for the region as a whole) (Table 5).

Table 5. Cumulative exposure indicator of future changes in fisheries (before normalisation of values). Values in fourth and fifth columns are the indicator scores (i.e., the relative change in landings scaled by proportional value of the individual species). For integration into the risk index, the exposure scores were reversed and then normalised to score between 0 and 1, so that a high score implied a high exposure to OA and climate change.

Management region	**Exposed provinces**	**Climate scenario**	**Exposure indicator scores**	
			2050	**2090**
Maritime	NS-mar; NB-mar	RCP 2.6	-0.114	-0.166
		RCP 8.5	-0.202	-0.498
Gulf	NS-gulf; NB-gulf; PEI; Que	RCP 2.6	0.042	0.017
		RCP 8.5	0.070	0.033
Newfoundland and Labrador	NL	RCP 2.6	-0.012	-0.026
		RCP 8.5	0.015	0.060
Regional total	n/a	RCP 2.6	-0.014	-0.034
		RCP 8.5	-0.018	-0.069

The Maritime management area is projected to experience losses for its quartet of exposures with the 2090 time-step and RCP 8.5 climate scenario resulting in losses of nearly 50% (Table 5). These scores were relevant to the NB-mar and NS-mar provincial sub-sections of NB and NS. Conversely, the Gulf area scored slightly positive in all scenarios, however 2090 for both climate scenarios had slightly reduced scores compared to 2050 (Table 5). The Gulf management area scores were applied to PEI and Que as well as the Gulf sub-sections of NB and NS. Interestingly, the Newfoundland and Labrador management area scored (slightly) negative under the RCP 2.6 scenario for both time-steps, but had positive scores under the RCP 8.5 scenario, largely due to the stronger gains in shrimp for the area under the latter treatment (Table 5). The Newfoundland and Labrador management area was only applied to the province of NL. The score as calculated for the whole Atlantic region was slightly negative under the quartet of treatments (Table 5).

3.3.3.2 Social vulnerability

The provincial scores for each of the indicators that contributed to vulnerability are presented in Table 6 and Table 7. The scores are presented in these tables as both the raw scores before normalisation of the data, as well as the normalised indicator scores which were used to calculate the final risk scores. The cumulative vulnerability scores (i.e., combinations of sensitivity and adaptive capacity) are presented in the final risk index in Table 8.

Quebec scored very low in nearly all of the indicators contributing to vulnerability (Table 6; Table 7). The only exception was the aquaculture indicator under adaptive capacity where, due to a relatively small shellfish aquaculture sector, Quebec scored moderately (Table 7). These factors likely stem from the social structure of Quebec being unique among the assessed provinces in many respects. Industry in the province is less dependent on natural resources and its population is an order of magnitude larger than any other province in the assessment. In fact, the population of Quebec is more than three times larger than the four other provinces combined. While at the same time, Quebec has the smallest fisheries sector of the five provinces.

Table 6. Indicators for sensitivity. Columns 2-3 present absolute scores for each province while columns 4-5 present the corresponding normalised scores as used to calculate provincial sensitivity (column 6). Higher sensitivity scores contribute to higher risk. The sensitivity scores were combined with the adaptive capacity scores (as indicated in Figure 11) in Table 7 to define vulnerability for each province (Table 8).

Province	Shellfish harvesting employment / capita(1)	Shellfish value / GDP(2)	Normalised shellfish employment score	Normalised shellfish value score	Sensitivity
NB	0.013	0.008	0.21	0.18	0.19
NL	0.044	0.019	0.76	0.48	0.62
NS	0.021	0.020	0.36	0.50	0.43
PEI	0.058	0.040	1.00	1.00	1.00
Que	0.001	0.001	0.00	0.00	0.00
Data references: (1) (L. Brander & Burke, 1994; DFO, 2002, 2007, 2015, 2017b; Gardner Pinfold Consulting, 2006a, 2006b; Stevens et al., n.d.); (2) (DFO, 2017f; Statistics Canada, 2017c)					

Table 7. Indicators for adaptive capacity. Columns 2-4 present absolute scores for each indicator. Normalised indicators were reversed (i.e., subtracted from 1.0) (columns 5-7), so that low indicator scores contributed to higher risk. The combined adaptive capacity score (last column) is the average of the reversed, normalised indicators. Adaptive capacity scores were combined with the sensitivity scores (as indicated in Figure 11) in Table 6 to define vulnerability for each province (Table 8).

Province	[*Reciprocal*] Un-employment / national un-employment(1)	Post-secondary education / national post-secondary education(2)	Aquaculture development / mollusc production(3)	(Reversed) Normalised un-employment score	(Reversed) Normalised education score	(Reversed) Normalised aquaculture score	(Reversed) Adaptive capacity
NB	0.635	0.888	0.262	0.25	1.00	1.00	0.75
NL	0.406	0.905	0.456	1.00	0.88	0.81	0.790
NS	0.676	1.024	1.297	0.11	0.00	0.00	0.04
PEI	0.529	0.940	1.108	0.60	0.62	0.18	0.47
Que	0.710	1.000	0.637	0.00	0.18	0.64	0.27
Data reference: (1)(Statistics Canada, 2017a); (2) (Statistics Canada, 2017b); (3) (DFO, 2017a, 2017f)							

NS had the second lowest cumulative vulnerability score (Table 8), largely driven by strong adaptive capacity indicators (especially aquaculture and education) (Table 7). Despite having the lowest education and aquaculture indicator scores, and a consequently low adaptive capacity, NB ranked third in vulnerability due to a relatively low reliance on shellfish, and hence a low sensitivity score (Table 6;Table 7).

PEI and NL were the most socially vulnerable provinces (Table 8). PEI's vulnerability derived from having the highest sensitivity (Table 6) due to a comparatively small population with a relatively high dependence on shellfish fisheries. Additionally, PEI earned moderate scores for two of the three adaptive capacity indicators, with relatively low education and high unemployment (Table 7). The strong aquaculture indicator score in PEI was not sufficient to offset the other indicators. NL had the lowest overall vulnerability score (Table 8). However, the only indicator where it earned the lowest score was unemployment (Table 7). The low overall vulnerability score was driven by generally weak scores in all the indicators, with an absence of any strong adaptive capacity or low sensitivity indicators to counterbalance the low scores.

3.3.3.3 Risk Index Scores

Risk from OA and climate change is anticipated where high exposure and high vulnerability coincide (Cardona et al., 2012; Mathis et al., 2015; Turner et al., 2003)

(Table 8). Since there was only a single vulnerability score for each province, while the exposure was represented by a quartet of scores, the patterns within a given province's quartet of risk scores mirrored the patterns in the exposure: the 2090 time-step generally had higher risk than the 2050 time-step. Under the RCP 2.6 scenario the risk scores tended to be higher due to more prominent gains in landings (and hence lower risk) projected under RCP 8.5. The risk scores coinciding with exposure to the Maritime management area (NB-mar and NS-mar) were the exceptions to this and had much higher risk under RCP 8.5 (Table 8).

Table 8. Risk index components and scores. Components are presented as final normalised scores. The exposure scores are presented in columns 3-4 as quartets of climate scenario and future time-step. The vulnerability scores (column 5) are the average (i.e., equally weighted combination) of sensitivity (Table 6) and adaptive capacity (Table 7) as indicated by Figure 11. The final risk score quartets (column 6-7) are the average (i.e., equally weighted combination) of the relevant exposures and vulnerabilities. The final two columns indicate rank for each unique risk score: 1-high, 2-moderate, 3-low, 4-minimal and 5-least risk (all categories except 3-low, represent 6 unique scores – 3-low only represents 4 unique scores).

Province	RCP	Exposure 2050	Exposure 2090	Vulnerability	Risk 2050	Risk 2090	Risk Ranking	
NB-gulf	2.6	0.049	0.092	0.471	0.26	0.28	4	3
	8.5	0.000	0.063		0.24	0.27	4	4
NB-mar	2.6	0.324	0.416		0.40	0.44	2	1
	8.5	0.479	1.000		0.48	0.74	1	1
NL	2.6	0.146	0.169	0.757	0.45	0.46	1	1
	8.5	0.097	0.019		0.43	0.39	2	2
NS-gulf	2.6	0.049	0.092	0.234	0.14	0.16	5	4
	8.5	0.000	0.063		0.12	0.15	5	4
NS-mar	2.6	0.324	0.416		0.28	0.33	4	3
	8.5	0.479	1.000		0.36	0.62	3	1
PEI	2.6	0.049	0.092	0.734	0.39	0.41	2	2
	8.5	0.000	0.063		0.37	0.40	3	2
Que	2.6	0.049	0.092	0.136	0.09	0.11	5	5
	8.5	0.000	0.063		0.07	0.10	5	5

Relatively high exposure scores were only achieved for the two provinces (NS and NB) that have fisheries in the Maritime management area. NS had fairly low vulnerability scores, while NB scored mid-range in vulnerability (Table 8). The highest single exposure score arose in the 2090 time-step under the RCP 8.5 climate scenario (Table 5;

Table 8). The effect of this exposure was strong enough that NB-mar and NS-mar scored the two highest risk scores. However, the low vulnerability in NS offset the exposure for the remaining Maritime management area quartet, and the remainder of the NS-mar risk scores were ranked low to minimal risk (Table 8). The higher vulnerability in NB meant the province was more influenced by the exposure indicator and consequently ranked high risk in three of the four of exposure scenarios (Table 8). Due to the occurrence of high vulnerability and moderate exposure, the NL risk quartet ranked high to moderate, with the RCP 2.6 climate scenario exposures scoring higher than the unmitigated scenario (RCP 8.5) (Table 8).

The remaining provincial units (NB-gulf, NS-gulf, PEI, and Que) are adjacent to the Gulf management area and therefore had very low normalised exposure scores (Table 8). PEI mainly ranked in the moderate risk category due to high vulnerability, with only the 2050 – RCP 8.5 exposure ranking as low risk (Table 8). The gulf sub-divisions of NB and NS (NB-gulf and NS-gulf) ranked from low to least risk. Out of the two, NB-gulf had slightly higher overall risk due to higher social vulnerability. Quebec, which had the lowest vulnerability along with the low exposures associated with the Gulf management area, predictably earned the lowest risk scores (Table 8).

3.3.3.4 Sensitivity testing

In order to test the robustness of the framework construction (i.e., Figure 11), alternate indicator weighting and indicator aggregating forms were considered. Most of the alternate orientations mainly re-arranged the mid-level risk values (i.e., the moderate to minimal risk categories). Although there were some notable adjustments in the higher risk categories. In all the tested weighting orientations, Que maintained the lowest risk scores.

Weighting each individual indicator equally (i.e., setting each indicator to contribute equally to total risk) rather than equal within each branch in the framework, caused the effect of the exposure indicator to be diluted so that the total risk scores were more similar to the provincial vulnerability scores. In this weighting system NL and PEI scored the highest risk index scores, followed by NB (Appendix 5). NS-mar and NS-gulf filled out the minimal and least risk categories.

When exposure, sensitivity and adaptive capacity were weighted equally (i.e., sensitivity and adaptive capacity were kept separate, rather than combined to form vulnerability), in a framework configuration which more closely resembled the orientations used in vulnerability assessments such as Ekstrom et al. (2015) and Cooley et al. (2012). NL and PEI again ranked higher due to their weak scores in the sensitivity and adaptive capacity (Appendix 5). In contrast, other risk assessment frameworks have used hazard, exposure and vulnerability as three equally weighted components of risk (Heinrich & Krause, 2016; Mathis et al., 2015). However, in this assessment, the hazard was effectively incorporated into the exposure term. To test the effect of including an equally weighted hazard component, the exposure scores were weighted twice as heavily as vulnerability. Under this weighting system NS-mar and NB-mar scored higher with some of their scores moving up a rank. Conversely, NL along with the NS-gulf and NB-gulf scored relatively lower (Appendix 5). Since the risk indices resulting from the alternate orientations emphasised either the vulnerability or the exposure indicators, the equal weighting system used in throughout the analysis presented the most neutral approach.

3.4 Discussion

3.4.1 Fisheries Changes in Atlantic Canada

Temperature-driven changes are expected to shift marine species' ranges poleward (e.g., Cheung et al., 2010; Cheung, Lam, et al., 2008; Perry, Low, Ellis, & Reynolds, 2005). With six of the seven species considered in this study currently near the middle or even at the southern extent of their historic natural range in Atlantic Canada (i.e., American Lobster, northern shrimp, snow crab, sea scallop, eastern blue oyster and Stimpson's surf calm (Palomares & Pauly, 2017)), it was unsurprising that the DBEM predicted changes in distribution in the study region resulted in the southernmost management area (i.e., Maritime) experiencing declines for most species under the quartet of exposure scenarios (Figure 13). Meanwhile, the Gulf and the Newfoundland and Labrador management areas were predicted to see modest increases in landings for most species under most of the treatments (Figure 13). However, snow crab, which requires very cold bottom waters, was predicted to see substantial declines in all scenarios and management areas (Figure 14c). This was consistent with other recent assessments of marine species distributions in Atlantic Canadian waters under warming ocean temperatures (Stortini, Shackell, Tyedmers, & Beazley, 2015). Due to its relatively high current contribution (both in value and tonnage) to overall fisheries landings in Atlantic Canada, the region-wide projected decline in snow crab impacted the cumulative fisheries changes for most of the management areas (Figure 13).

American oyster was the only assessed species currently near the northern limits of its current distribution around NS and NB (i.e., in the Maritime management region). Present landings for the species are relatively small (Figure 12; Figure 13), but overall, the region is expected to become better suited for the species. With its suitability for aquaculture production, enhanced production of the species could help to mitigate losses anticipated of other species, especially in the southern reaches of the region. However, current American oyster production rates are orders of magnitude lower than most of the other assessed species, so any mitigation potential for losses are likely to be highly localised.

3.4.2 Risk and Social Vulnerability

The communities in NB that rely on fisheries in the Maritime management area were found to be most at risk to OA and climate change, even though the bulk of NB shellfish production occurs in the Gulf management area and is not exposed to the changes in the Maritime management area. Nonetheless, this sub-section of the province should be considered as an area for pro-active responses to potential changes in fisheries driven by OA and climate change. Conversely, the NB-gulf sub-section did not appear to be a high risk area, and scored minimal risk in all of four of the exposure scenarios. The contrasting risk levels within the province may present an opportunity to locally mitigate the risk in the Maritime management area dependent communities, as production and harvesting activity could be shifted to the potentially less exposed management area. But, shifts such as this would certainly have to take the potential consequences of increasing fishing capacity in the Gulf management area into account.

The provinces of PEI and NL were also at higher risk from OA and climate change, with most of their scores in the high to moderate risk categories. The risk posed to these provinces was predominantly driven by higher social vulnerability. Many of the vulnerability factors (such as education and unemployment) occur at local scales and are therefore more immediately actionable by provincial decision-makers. Therefore, these provinces may be able to more directly pursue social and economic shifts to reduce risk from OA and climate change.

Overall, NS scored a single high risk score (in 2090 under RCP 8.5) for the Maritime management area; otherwise the province ranked low to least risk. In spite of being highly exposed in communities which depend on the Maritime management area. Nova Scotia's relatively low vulnerability suggests that the province will, generally, be better able to respond to the changes in shellfish production driven by OA and climate change than most other provinces in the region. However, it is possible that communities within the province have much less capacity to adapt than the provincial statistics imply. As with most of the other provinces in the region, NS has a relatively high rural population and many small coastal communities with a high dependence on fisheries (DFO, 2004; Statistics Canada, 2011). Given the relatively strong declines anticipated for the Maritime management area, a finer grained investigation may be warranted for this province.

Lastly, Que appears to be largely unthreatened by OA and climate change impacts on fisheries. The province as a whole is not vulnerable to changes in shellfish production, and under the DBEM projections the comparatively small shellfish harvest was not highly exposed to change.

3.4.3 Responding To Risk

To reduce exposure to OA and climate change action at global scale required. Although some of the local amplifiers of OA (e.g., eutrophication) can be acted upon at more localised scales (Kelly et al., 2011). In order to reduce risks to Atlantic Canada (and the rest of the planet) posed by OA and climate change, global efforts need to be made to reduce carbon emissions (e.g., Doney et al., 2009; Hoegh-Guldberg et al., 2014; IPCC, 2014). Local policies can be enacted to reduce emissions and contribute to global reductions in emissions. However, the tangible global benefits of reduced emissions in Atlantic Canada will be relatively minor.

Factors affecting social vulnerability can be acted upon locally by regional managers and decision-makers to reduce risk. Although this will not reduce the damage caused by OA and climate change, it can reduce the effects felt by human communities. With respect to vulnerability related to Atlantic Canadian shellfish fisheries, there are multiple opportunities for mitigation actions. While education levels were near the national rate for most of the provinces (Table 7), efforts to improve education rates could alleviate some of the social vulnerability in the region by opening more opportunities to the populations. The education statistic considered here was a broad indicator of overall education in the provinces, however, more targeted education programs regarding the future of the affected fisheries (for both increasing and decreasing abundances) could greatly improve the adaptability of the harvesters and communities which rely on them and help to promote sustainable long-term harvests (Madin et al., 2012). Similarly, addressing unemployment rates, which were above the national levels in all of the provinces, could greatly reduce vulnerability to potential lost employment in declining fisheries. Improvements in either (or both) of these indicators should also improve provincial adaptive capacity regarding a range of potential climate change driven impacts.

The third adaptive capacity indicator, aquaculture, is more case specific but also presents a direct way to respond to OA impacts on shellfish production in the region. Current aquaculture production in Atlantic Canada is mainly dependent on wild populations because hatchery production in the region is limited (Isabelle Tremblay, personal communication, November, 03, 2017). However, investment in hatchery infrastructure could greatly improve the region's ability to maintain production of mollusc species in the face of falling pH or irregular natural recruitment. Adaptation of hatchery procedures is already being used to mitigate low pH events on the West Coast of Canada and the United States; the expertise developed there could be leveraged to support Atlantic production (Clements & Chopin, 2016; Washington State, 2012). It is worth noting that while improving shellfish aquaculture may help to maintain production levels, it is not a direct substitute for loss of livelihood from decreased capture fisheries. Fish harvesters are unlikely to actively pursue a transition to aquaculture production. Furthermore, in terms of actual employment aquaculture is more efficient and requires fewer people to produce higher harvests. Nonetheless, as a factor to reduce overall risk from changes in fisheries, aquaculture (with hatcheries) is an opportunity worth further consideration in the region.

The indicators related to sensitivity are somewhat more difficult to address because they are more directly tied to the fishing industry and are ingrained in the social and cultural identity of the region. Furthermore, DFO acknowledges that overcapacity already exists in many Atlantic Canadian fisheries (DFO, 2004). Reducing overall fishing effort could help to limit sensitivity to changes in the fisheries and support overall conservation targets (DFO, 2004; Pauly et al., 2002). To be a viable option, this would first require that alternative employment opportunities existed. Diversifying harvests is another mechanism that can be enacted to dampen losses in any one fishery (DFO, 2004). Ekstrom et al. (2015) considered the diversity of mollusc species harvested as a component of adaptive capacity in their assessment of vulnerability to OA in the United States. However, many of the commercial harvesters in Atlantic Canada already target multiple species (DFO, 2004). While this may ultimately indicate that the provinces are less sensitive to changes than this assessment implies, it also means that this aspect will be difficult to improve upon as a method to reduce vulnerability for the region. Developing other industries and factors supporting provincial economies could reduce the relative importance of fisheries in the region and thereby reduce sensitivity related to changing fisheries.

A finer scale assessment is also potentially highly relevant for addressing risk in Atlantic Canada. Much of the region is made up of small communities with much more localised economies than is represented by the provincial statistics (DFO, 2004; Statistics Canada, 2011). In some counties, employment in shellfish fisheries is much higher than the provincial rates imply – especially when additional steps of the supply chain are included. For example, lobster fishing area (LFA) 34 is by far the most productive area for American lobster in the Maritime management area (Tremblay, Pezzack, & Gaudette, 2012). Following the approximation that counties rely most heavily on adjacent management areas, this implies that Digby, Yarmouth and Shelburne counties could be much more sensitive to changes in that fishery than other NS counties. Furthermore, LFA 34 corresponds to the area with some of the most substantial DBEM projected losses for lobster landings (Appendix 3 – Figure A3.1-e-h). Future assessments at a finer scale might be necessary to address these potentially more at-risk communities. Following the conclusions of this study, NB and NS would be strong candidates for future fine-scale assessment, as these provinces were most exposed to changes in fisheries. Therefore highly vulnerable communities in these provinces could be among the most at risk in the region.

3.4.4 Conclusion

The findings of this assessment contrast with previous socioeconomic analyses of OA effects on fisheries. In other assessments, projected changes in fisheries have been presented almost exclusively as declines in potential landings with subsequent impacts on societies and economies. However, the majority of these studies did not account for effects from other climate change factors (e.g., temperature). This study demonstrates that accounting for other environmental factors can allow for different, and potentially more representative, narratives to emerge. Future management decisions and mitigation plans could be better informed through analyses which account for a more complete range of future effects on resource accessibility.

Atlantic Canada is a region with an exceptionally high dependence on OA-susceptible fisheries. Nonetheless, the findings presented here suggest that OA-driven declines in the region will be minor compared to temperature-driven changes in future potential landings (Figure 15). Fisheries resources in Atlantic Canada are expected to see some notable redistributions under OA and climate change over the coming century. As observed in global-scale studies this is expected to result in 'winners and losers' with respect to access to future fisheries (e.g., Cheung et al., 2010). It may be tempting to view the 'winners' as gaining access to new production. But it is essential to bear in mind that perceived increases come at a cost to other regions (Mumby et al., 2017). Furthermore, OA effects may appear to be locally overwhelmed by temperature driven increases; but, this should be seen in the context of limiting potential gains, and exacerbating declines.

Within the Atlantic Canadian region, management plans should be developed that take climate change and shifting species distributions into account. These plans should specifically address future allocation of resources when long-held access to certain high value fisheries (e.g., American lobster) cross into new jurisdictions. Moreover, these considerations need to be extended to international management of marine resources. Any net gains in Atlantic Canada will be coming at a cost to American states to the south.

Management plans seeking to account for changing access to resources will benefit from assessments, such as this thesis, which address how and where biological changes are likely to affect human communities. A more robust scientific understanding of the complete and combined biological impacts of OA and climate change will allow for future socioeconomic assessments to better predict where changes in resources will be most relevant.

Chapter 4 Conclusion

4.1 Research Summary

Ocean acidification is an aspect of climate change that has gained traction and attention in scientific and public spheres in the past 10 to 15 years. Public interest in the phenomenon in North America was largely triggered by the significant economic losses driven by OA-related events which occurred in the Puget Sound area of Washington State between 2005 and 2009, and the resulting die-offs of oyster larvae in hatcheries (e.g., Grossman, 2011; Washington State, 2012; Welch, 2013; Xiong, 2016). These instances were driven by upwelling of low pH water from the deep ocean (a local amplifier of OA) and presented one of the first tangible examples of the potential damages that may arise from OA.

While a complete scientific understanding of the biological implications of OA is still developing (Kroeker et al., 2013, 2010, 2017), there is a need to understand how OA might affect human communities and economies (Hoegh-Guldberg et al., 2014; IPCC, 2014). To date, a limited number of studies have started to address how OA might affect societies (Chapter 2). Most of these consider how OA is likely to impact shellfish fisheries; although, there is also a small number of investigations into finfish fisheries as well as other ecosystem services (e.g., L. M. Brander et al., 2012; Voss et al., 2015).

Following a review of the methods used thus far to assess potential socioeconomic impacts from OA, this research set out to contribute to the growing body of literature by investigating the risk posed to Canada's Atlantic Provinces through OA and climate change driven shifts in distributions of seven high value shellfish fisheries (Chapter 3). The thesis successfully integrated results from a biophysical model and a social risk framework to evaluate the socioeconomic risk that Atlantic Canadian provinces will face over the 21st century. The analysis was built on predictions from a DBEM, which uses species distribution, population dynamics and climate -models to predict future distributions of marine species. Projected changes in potential landings for three DFO management areas were used to estimate how exposed dependent provinces will be in the middle (2050) and end (2090) of the 21st century, under two different CO_2 emission trajectories (RCP 2.6 and RCP 8.5). The social vulnerability of each province was determined based on five indicators designed to represent sensitivity to changes in the fisheries and the broader socioeconomic adaptive capacity of the provinces.

Previous research in this domain has linked biophysical models (of varying complexity) with narrower economic impact estimates, or has assessed the risk without directly modelling changes in relevant fisheries. Combining an explicit biophysical model with a risk assessment allowed the social risk to be informed specifically by expected changes in access to the resource of concern. This represents a step forward to assessing the susceptibility of human communities to OA and climate change impacts on fisheries resources.

4.2 Main Findings

4.2.1 Current Literature on Socioeconomic Impacts of Ocean Acidification

To date eleven studies have attempted to identify the social and economic implications of OA related changes in shellfish fisheries. Within these studies two main methodological approaches were followed. In some, economic analyses sought to quantify the change in revenue or value for specific changes in fisheries landings. In others, social vulnerability or risk assessments identified communities where OA driven changes are most likely to have socioeconomic impacts.

Both methods have relevance for responding to OA. The economic analyses can help to contribute to quantifying the total cost of anthropogenic CO_2 emissions, although they do not identify how or where in a society the impacts are most likely to be experienced. Thus the outcomes of economic analyses can be useful for global policy decisions related to climate change mitigation as they allow real dollar values and well-defined scenarios to inform decisions. Conversely, the SVR assessments tend to avoid making explicit estimates of changes, but are very useful for highlighting where the effects are likely to be most severe and where mitigation actions should be prioritised. Therefore, these assessments are potentially more useful for local decision-makers to manage resources effectively and plan for responding to future impacts.

4.2.2 Risk from Ocean Acidification and Climate Change in Atlantic Canada

According to the DBEM projections, Atlantic Canada is expected to experience re-distributions of the seven assessed shellfish species. Most of the assessed species exhibit a general northward shift through to 2090. This shift was particularly evident under the RCP 8.5 emissions climate scenario (Figure 14; Appendix 3). When all species were considered together, the cumulative changes across the region as a whole were minimal, with a net change in combined landing weights of less than 4% anticipated under RCP 8.5 (less than 2% under RCP 2.6). However, on a species by species basis, some species were expected to decline across the whole region (e.g., snow crab), while other saw some localised declines which were more than offset by gains elsewhere within the region (e.g., northern shrimp) (Figure 13). When considered in isolation, OA was anticipated to have a negative impact on future availabilities of the seven species. However, the OA signal was consistently overwhelmed by temperature effects which drove the main direction and level of impact for each of the assessed species across the region (Figure 15).

The DFO management area in which landings were projected to decline most consistently (Maritime) is adjacent to two provinces, NB and NS, with moderate to low relative vulnerabilities to changes in fisheries. The provincial vulnerabilities were driven by comparatively low reliance on shellfish fisheries in NB, while NS scored strongly in factors which contributed to adaptive capacity (such as high education and low unemployment). Conversely, management areas where landings were expected to increase or remain fairly stable (Gulf, and Newfoundland and Labrador) are adjacent to provinces with higher relative social vulnerability (i.e., NL and PEI – Que was also adjacent to the Gulf management area but had the lowest vulnerability to changes in the assessed fisheries). In PEI the vulnerability was mainly due to relatively higher employment and economic value derived from shellfish fisheries. Vulnerability in NL was imposed by weak scores for most of the general social factors that contributed to adaptive capacity.

This inversed relationship between expected exposure and vulnerability in the region suggests that Atlantic Canada may be at less risk from OA and climate change than originally hypothesised. Note that, due to the situation-specific nature of risk assessment frameworks, the findings of this regional analysis are only directly relevant within the region. To better assess the risk posed to the region as a whole, a broader-scaled analysis would be necessary. If Atlantic Canada was placed within an assessment at a larger social and geographic scale, the findings of this analysis would be strengthened as they could be related to a more comprehensive picture of social and economic risk posed by OA and climate change. Similarly, assessments at finer social scales for the individual provinces which appear to be most at risk (i.e. NB, NL and PEI – Table 8) could identify specific communities which might be at greater risk than this regional scale analysis implies.

4.3 Challenges, Limitations and Assumptions

Many of the assumptions that were required to proceed with the analysis arose from the highly interdisciplinary nature of the assessment which required bringing together many data types from different sources with inherently different scales and intended applications. In all cases assumptions were made in a manner that attempted to follow the most straightforward and neutral option possible given the available data.

4.3.1 Fisheries and Oceans Canada Data

Canada has strong marine management capacity and data reporting (at least since establishment of its EEZ in 1977) that is generally seen as reliable (Divovich et al., 2015; SAUP, 2006). Nonetheless, there were aspects of the available data that were lacking or presented challenges. First, the data used to define the baseline of the analysis was an average of annual landings from 1991 to 2010; this time-frame was selected to align with the 20 year running averages used in the DBEM outputs. However, it overlaps with a period of significant change in the ecological community in Atlantic Canadian waters. The groundfish fishery collapsed in the early 1990s and resulted in a rapid rise in shellfish (mainly crustacean) populations, due to predator release (i.e., Atlantic cod and other groundfish are natural predators of crustaceans whose decreased abundance allowed crustaceans to thrive), and subsequently led to increased harvests (Dawe et al., 2012; Divovich et al., 2015). Following this shift, the profile of the landings at the start and the end of the baseline period are different. A more temporally constrained average annual landing weight (i.e., from 1996 to 2005), was found to be slightly higher than the 20 year period average used in the analysis. Since these options were relatively similar the 20 year average of the DFO data was used in the analysis as it aligned with the 20 year running means which were used to smooth the DBEM outputs.

It was assumed that provinces relied exclusively on the adjacent management areas and did not harvest in other management areas. For most species this is likely accurate; however some offshore fleets certainly operate in management areas which are not adjacent to their province (e.g., NS harvesters hold significant quotas for northern shrimp in the Newfoundland and Labrador management area). Lacking specific compositions of fleets or licence specific landings, this was the most straightforward assumption to follow.

Nutritional and subsistence importance of the fisheries was also excluded from this research. Data related to this aspect of fisheries in Atlantic Canada is not widely available (Berkes, 1990; Divovich et al., 2015), and was therefore interpreted as representing a minor component of the overall importance to the provinces. However, it is likely that in some communities and households, locally harvested shellfish represent a critical resource, and this topic may warrant further investigation. Similarly, data on cultural importance, especially for First Nations communities, was not identified but may be a significant locally important aspect of shellfish harvest. These types of data would have allowed for a more wide-ranging view of the vulnerability of the assessed provinces. Given the expected small fractions of the provincial populations for which these would have been relevant, it is unlikely that the overall findings of this assessment would have been affected. If this assumption is incorrect and subsistence fisheries are in fact highly important to significant portions of any of the provinces, then the vulnerabilities of the respective provinces could be much higher than assessed. On a smaller, within-province scale, where local importance could be emphasised these data would be much more relevant.

4.3.2 Challenges Associated with Using the Dynamic Bioclimate Envelope Model

Challenges related to scale: The DBEM is a large, highly complex and detailed model. Like all models it has trade-offs regarding when, where and at what scales it is most reliable. The DBEM uses earth system models as underlying drivers of change, and thus uses a relatively coarse (half-degree latitude by half-degree longitude) spatial scale.

Furthermore, global circulation models often cannot account for localised currents and processes in coastal systems. Coastal systems are further complicated for OA and pH due to the high variability and fine scale effects the coastal environment has on the marine carbon system (Doney, 2010; Duarte et al., 2009). Therefore, caution should be exercised when interpreting the DBEM outputs at the comparatively small scale of Atlantic Canada. This was made particularly evident in the Gulf management area with the snow crab data where the baseline DBEM distribution significantly under-represented the DFO reported landings (Appendix 4). Given the complex coastal systems involved in Atlantic Canadian waters, it was not surprising that the DBEM distributions were rather imprecise when directly compared against DFO reported landings (Appendix 4). This disconnect may also be partially explained by the previously mentioned shift in ecosystem structure in the region in the early 1990s. Since this was driven by overfishing and not natural shifts in habitat suitability, it may have affected the DBEM determination of baseline (year 2000) which is inferred from the preceding five decades.

The DBEM projections are intended to present long-term trends rather than interannual variability. Previous research using the DBEM has determined that projections reflect longer term average catches (i.e., smoothed catch data) reliably. In a case study with sablefish in Alaska, Cheung et al. (2016) showed that the 20-year mean of real landings fell within the range of DBEM predicted landings. Furthermore, previous investigations of species distribution shifts with the DBEM have concluded that the outputs were reliable for similar latitudes to Atlantic Canada (Cheung et al., 2010; Cheung, Sarmiento, et al., 2013).

For the purposes of this assessment, it was assumed that while the baseline DBEM distributions were somewhat misaligned with respect to the DFO landings, the relative changes in distributions, as predicted by the DBEM, were reliable. Effectively, under changing ocean conditions, the suitability of the habitat (with respect to temperature, depth salinity and habitat type) was expected to change as predicted by the model. Therefore, the relative changes predicted by the DBEM were seen as applicable to the DFO landings for this first order analysis.

Integration of ocean acidification into the model: OA may interact with other climate stressors more strongly than the model allows for, again resulting in potentially underestimated impacts. Ocean acidification and other stressors may synergistically interact, as seen in laboratory studies, so that resulting impacts in nature may be more severe than those modelled (Kroeker et al., 2017; Portner, 2012). The DBEM includes OA as an impact on both growth and survival. The impacts on growth interact with oxygen demands and temperature changes. In this way the DBEM accounts for a theoretical interaction between OA and other climate stressors. The survival impacts in the model do not interact with other stressors (Tai et al., In prep.). In their analysis based on DBEM projections Lam et al. (2014) suggested their results regarding the OA impacts were likely conservative. The current inclusion of OA in the DBEM is likely an over-simplification, but it presents a possible pathway for interactions between OA and other stressors which is currently lacking in most socioeconomic assessments of OA impacts.

Due to the current unevenness of data regarding species-specific response to OA, the DBEM uses highly generalised OA effects that are only differentiated among very broad taxonomic groups (i.e., molluscs and crustaceans), based on the meta-analyses conducted by Kroeker et al. (2013, 2010). For most species the applied OA impacts are therefore likely to be inaccurate. Given the lack of species-specific response data the meta-analysis impacts are, currently, the best available approximation for examining a broad a range of species. Future work with the DBEM could include more specific OA responses for each species and also account for differential impacts across life-cycle stages.

Relatedly, the OA impact level used in this iteration of the DBEM was the mean effect from the Kroeker et al. (2013; 2010) meta-analyses. The OA driven losses could be much more severe than anticipated by the model, potentially even to the point where OA might not be entirely overwhelmed by temperature driven patterns in abundance. Conversely, the OA impacts could also be much less severe and the benefits for many species in the region from thermal loading could be more pronounced. In their economic analysis of OA impacts Narita and Rehdanz (2016) compared the upper and lower OA effect levels from Kroeker et al. (2013). Tai et al. (in prep.) are testing different impact levels if OA in the DBEM. Thus future socioeconomic assessments using DBEM projections may be able to examine a broader range of OA impact levels.

The OA effects in the iteration of the DBEM outputs used in the thesis also assume a linear impact from OA (i.e., as pH changes there was a linear relationship to the affected life-history traits – growth and survival). This treatment does not allow for adaptation, where species might be able to limit the effects of OA through biological mechanisms such as acclimatisation (including behavioural responses), parental effects (i.e., epigenetic effects) or evolution (e.g., Branch et al., 2013). Tai et al. (In prep.) indicate that including a simulation of adaptation in the DBEM (where the pH level is applied in relation to the previous time-step rather than the baseline level) leads to negligible effects from OA. In the assessment of Atlantic Canadian fisheries, a reduced impact from OA would have resulted in more pronounced increases (e.g., American oyster) and would have reduced the declines observed for some of the other species (e.g., sea scallop) (Figure 15).

Conversely the modelled impacts also did not allow for a threshold effect, where below a certain pH, a species cannot persist at all. Again, Tai et al., (In prep.) are investigating how threshold effects might alter OA impacts in the DBEM. In testing for a threshold, or tipping point, Tai et al. applied an exponential relationship between pH and the OA impacts (growth and survival). This resulted in low impacts in the early time-steps, but much larger effects by the end of the century. If similar OA effects were included in the risk assessment for Atlantic Canada it is possible that the exposure terms would have been much stronger across the region, especially for the 2090 time-step. However, at the time of this assessment, these alternative methods of integrating OA into the DBEM were not available. Therefore, outputs based on the mean impacts on growth and survival from Kroeker et al., (2010, 2013) were the most relevant biophysical model outputs that were available to inform the risk framework.

4.3.3 Social Limitations and Assumptions

A significant limitation inherent in the risk and vulnerability assessment approaches is that the outcomes are not directly comparable to other assessments due to the specificity with which assessment frameworks are designed (Cardona et al., 2012; Turner et al.

2003). Since all of the indicator scores were normalised between the maximum and minimum observed values for each specific indicator, the final index scores are only indicative of the risk relative to the other units (i.e., provinces) in the assessment. Therefore, the results of this analysis are not directly comparable to the results obtained in other similar analyses – although, as suggested by Turner et al. (2003), patterns and commonalities driving high and low risk scores can potentially be considered. Furthermore, the conclusions represent a potentially very useful tool for decision-making within the study region, since understanding where exposure to changes in fisheries is most likely to affect communities can allow for proactive action either in managing access to resources or addressing wider social vulnerabilities.

Many important assumptions regarding the quantification of social factors in the assessment were also necessary to proceed with this analysis. A key assumption related to the social data was that each province was internally uniform with respect to the social attributes used as indicators. This is unlikely to be the case, as there are substantial differences in the demographics and socioeconomic status between urban and rural communities in the region. Approximately 50% of the population in the Atlantic Provinces (not including Quebec) live in rural areas, whereas the national rate is closer to 20% (Statistics Canada, 2011). Combined with the acknowledged importance of fisheries to coastal communities (DFO, 2004), this implies that provincial-level statistics may not be representative of finer-grained patterns in the social structure of these provinces. Schmidtlein et al. (2008) determined that vulnerability between assessment scales was consistent (i.e., that coarser scale areas with vulnerability were made up of finer scale areas also with generally higher vulnerability). However, within their case studies there were some examples of highly vulnerable units at the finest scale which were incorporated under moderately vulnerable aggregate areas. It seems likely that similar patterns could exist in Atlantic Canada. In general the provincial risk scores may represent their component communities, but this scale might overlook communities with particularly high risk. As a preliminary assessment of Atlantic Canadian risk, provincial units are, nonetheless, a good starting point for addressing general patterns.

It is highly probable that the relative value of the fisheries will change through time as market demands and consumer preferences change. For example, American lobster was used to feed prisoners and servants in colonial America, but it is now among the most valuable fisheries (DFO, 2017c; GMRI, 2012). Economic value of harvests is especially likely to be affected if changes in future production change. Given that changes in production are the foundation of assessments such as this thesis, the potential for changes in value should be considered. This assessment attempted to avoid making a specific estimate of future landed values, and instead focused on the current value of the fisheries. However, by not making predictions of value change it was implicitly assumed that relative value among species will be constant through time. Due to the different patterns in composition of landings between provinces it is possible that changes in relative value between species could affect the outcomes of the risk framework. However, estimates regarding the changes in relative value would have been tenuous and likely would not have substantially altered the findings of the assessment.

Employment in fisheries was also difficult to quantify. Since most of the fisheries in Atlantic Canada are seasonal it is probable that many active harvesters operate in more than one fishery and/or work in other sectors during the off-season (Michael Gardner, personal communication, September 11, 2017). In fact, fisheries managers support harvesters targeting diverse fisheries as it reduces their dependence on any single fishery and allows for better response to changing market demands (DFO, 2004). These factors mean that using licence numbers (multiplied by crew sizes) as an approximation of employment within specific fisheries may be over-representative/double-count employment numbers in some fisheries. Since the employment indicator in this assessment was only used to compare the provinces relative to each other, it was seen as reasonable first order approximation of primary harvesting employment rates between provinces (Michael Gardner, personal communication, September 11, 2017). Secondary employment in sectors related to fisheries (e.g., processing) was not included in this assessment because it would have been difficult to constrain to shellfish only operations. A more in depth assessment of the sensitivity to changes in shellfish production could seek to identify these more subtle factors and be used to show the broader importance of shellfish fisheries to local communities. Employment in the processing sector could also further differentiate importance of fisheries between provinces since the provincial ratios between primary harvest and processing employment are very different. For example, in NB roughly half of the total employment related to all fisheries occurs in processing, while in PEI only a quarter of total employment comes from processing (DFO, 2017d). This suggests that the broader social importance of employment from fisheries to the provinces could be different than the primary harvesting employment data implies. However, as previously indicated, processing employment data that differentiated between fishery types was not readily available for this assessment.

4.3.3.1 An Aside on Aquaculture

Aquaculture is a growing industry and undoubtedly has a role to play in the future of global food security (FAO, 2016). In terms of adapting to OA and climate change, aquaculture also presents a significant opportunity to maintain current levels of production of some susceptible species (Clements & Chopin, 2016). However, from a social perspective, aquaculture should not be seen as a straight substitute for past (or current) commercial harvest activities, as the employment potential is typically much lower per unit of production. In 2015, Canadian commercial harvest employed ~42,500 people and generated ~$3,260 million, while aquaculture employed 3,300 people and generated $967 million. Despite this order of magnitude difference, per-person value is much higher in aquaculture (DFO, 2017c). Furthermore, the lifestyle and personal identity associated with aquaculture employment is vastly different from the personal identity that is often associated with wild harvest activities. Nonetheless as an economic indicator regarding total production of shellfish species, aquaculture is a critical component to include in consideration of future shellfish production.

4.4 Future Management

If the DBEM-projected gains in the landings for the Gulf and the Newfoundland and Labrador management areas occur, they must be considered in the context of a re-distribution of landing opportunities rather than outright growth in production. This perspective would ensure that the declines anticipated for the Maritime region would not be overlooked by local decision-makers and future management plans could incorporate, and account for, responses in other management areas (Madin et al., 2012; Mumby et al., 2017). Moreover, this awareness would limit the tendency to interpret increasing stocks as increased productivity of the population and a reason to increase fishing effort (Mumby et al., 2017).

All marine fisheries in Canada fall under the jurisdiction of a single federal branch of government (i.e., DFO). This, in theory, should allow for effective management of changing resources and ultimately, harvesting opportunities, as they shift across provincial boundaries. Nonetheless, Atlantic Canada has a long history in fisheries and instances of non-compliance with management decisions and conflicts between harvesters are not uncommon (DFO, 2004; Divovich et al., 2015; McMullan & Perrier, 2002). These management issues are ongoing. A recent DFO external review panel regarding the 'last-in-first-out' (LIFO) policy for quota allocation in the offshore shrimp fishery in the Newfoundland and Labrador management area has raised tensions between provinces. NS harvesters, who were the first to enter the fishery, pushed for upholding LIFO, while harvesters in NL saw the resource as belonging to local operators and were strongly in favour of its annulment (Barry, 2016; Muligan, 2016; Sprout, Crann, Follett, & Taylor, 2016). However, past (and present) disputes amongst harvesters, and between harvesters and managers, in relation to (perceptions of) unjust allocation have all been in a (perceived) context of a stable resource base and an unchanging environment – that is, absent the influence of a strong directional pressure such as climate change and OA.

When strong directional pressures, such as climate change, impact future stocks it is possible that conflicts will worsen. The DFO has set out an extensive policy framework that indicates a desire for a less prescriptive and more collaborative management approach to encourage more collaborative management with harvesters with a further goal of ensuring conservation and sustainable use of Atlantic Canadian fisheries (DFO, 2004). However, there is no direct consideration of potential climate change-driven impacts on fisheries. The current policy framework acknowledges that uncertainty with respect to access to fishing rights has, in the past, been a strong contributing factor to non-compliance with management decisions (DFO, 2004). As such, ignoring drivers that have the potential to fundamentally change patterns of harvest opportunities seems short-sighted. Future policy should explicitly account for, and consider, climate change as a source of uncertainty that may not only affect stock availability, but also indirectly encourage non-compliance. If harvesters believe that fish stock distributions will change and their access to a given fishery could be lost, there may be little perceived reason to harvest sustainably (Madin et al., 2012; Mumby et al., 2017). A management plan that accounts for shifting stocks as climate change-related impacts unfold and allows for some level of continued access, or compensation for lost access, could encourage more sustainable use of resources in the long term by strengthening the compliance culture across fisheries and limiting the uncertainty with respect to access (Mumby et al., 2017).

Management challenges related to climate change-driven shifts in species distributions will undoubtedly extend beyond national borders. American lobster populations are already declining in the state of Maine, while landings in NS have increased (Greenhalgh, 2016; Wahle, Dellinger, Olszewski, & Jekielek, 2015). Canada and the United States have a long history of engaging in treaties and agreements regarding marine species with transboundary distributions (e.g., the Halibut Treaty (Hillmer & Scott, 2017) and the Pacific Salmon Treaty (DFO, 2017e)). Despite this long history there have been significant (and relatively recent) instances of non-cooperation between the two nations. Notably, in the early to mid-1990s when a shift in Pacific salmon (*Oncorhynchus spp.*) migration patterns in response to natural ocean cycles resulted in higher catches for Alaska, at a loss to Canadian interests, cooperative management collapsed (Miller, Munro, McDorman, Mckelvey, & Tyedmers, 2001). In response to failing stocks in the late 1990s, new agreements were made and cooperative management appears to have been re-established. However, Miller et al. (2001) caution that future changes in distribution of stocks across borders have the potential to destabilise the current balance.

Similar to management challenges between provinces, international agreements thus far have been made in the context of largely stable distributions and where each side is motivated to cooperate to avoid dangerous, mutually destructive harvesting strategies. As seen in Pacific salmon, the scenario changes dramatically when harvesting opportunities shift and motivations to cooperate erode (Miller et al., 2001). Nonetheless, the agreement regarding the interannually variable Pacific salmon stocks could provide a good starting point for a framework for international management of species in the midst of climate related redistributions such as American lobster. A key aspect that Miller et al. (2001) strongly supported was the indirect inclusion of side-payments to compensate for lost fishing opportunities. The strong cooperative approach seen in the Pacific salmon agreement could also be leveraged to foster cooperative management relations between Nations on the Atlantic Coast.

Ideally, resource managers across borders should proactively consider strategies to respond to shifting species and harvest distributions (Madin et al., 2012; Mumby et al., 2017). Assessments of future distributions of marine species and their socioeconomic impacts, such as this analysis, can assist in motivating and developing future transboundary management strategies in advance of conflict, as well as help to prepare resource users for potential changes in access to historic and potentially culturally significant fisheries.

4.5 Future Research

4.5.1 Ocean Acidification Research

As has been repeatedly recommended in previous socioeconomic assessments regarding potential effects of OA, a more robust scientific understanding of the biological impacts of OA will greatly improve predictions of its social implications (e.g., Cooley & Doney, 2009; Hilmi et al., 2013; Mathis et al., 2015; Narita et al., 2012). A better understanding of how OA and other climate change stressors (e.g., temperature) will interact will also be fundamental for more accurate projections of the future of fisheries (Crain et al., 2008; Kroeker et al., 2017). The current limits on understanding are especially relevant for finfish responses to changing pH. While there is evidence that at least some finfish will show behavioural and physiological responses to declining pH (e.g., Dixson et al., 2010; Munday et al., 2009; Murray et al., 2016), responses across species assessed to date are highly variable Kroeker et al. (2013, 2010).

Due to the overall lack of a thorough and complete understanding of OA effects, social assessments have largely avoided potential changes to finfish fisheries. Although there are exceptions, Mathis et al. (2015) and Heinrich and Krause (2016) both approximate a food-web impact on selected finfish in their assessments. Additionally, Voss et al. (2015) present an investigation of social impacts in relation to anticipated changes in a

Norwegian cod fishery. In order to fully address social impacts of OA a more complete understanding of potential impacts on finfish is essential. This represents a critical gap in the understanding of social and economic susceptibilities to OA since the majority of wild fisheries are derived from finfish species (FAO, 2016).

Relatedly, linking the biological effects with broader ecosystem impacts needs further investigation and understanding. If low trophic level species are heavily impacted there is a strong potential for declines to cascade throughout ecosystems, leading to major restructuring or complete collapse (Branch et al., 2013; Gaylord et al., 2015). Ocean acidification may also impose more subtle ecological impacts (e.g., altering a species' susceptibility to predation, or impacting ability to compete for food), and could affect which species are dominant within an ecosystem and ultimately alter ecosystem function and structure in less predictable ways (Gaylord et al., 2015; Kroeker et al., 2017). These effects could potentially affect fisheries even if target species are themselves largely unaffected. Ecosystem effects may be better addressed with models such as the Atlantis model (Fulton et al., 2004) which explicitly accounts for food-web relationships between species. However, this model would still benefit from greater understanding of the previously mentioned data gaps, especially an understanding of the more nuanced OA effects (K. N. Marshall et al., 2017).

4.5.2 Future Socioeconomic Investigations of Ocean Acidification and Atlantic Canada

Additional research targeting socioeconomic impacts of OA and climate change-driven shifts in Atlantic Canadian fisheries could help to support future management decisions for the region. However, much more tangible benefits would be gained through actual development of proactive management plans. The DFO has recognised the strong potential for OA to impact fisheries and ecosystems in Atlantic Canada (e.g., DFO, 2014). In contrast, however, there appears to be a lack of management policy to adapt to anticipated changes. While a complete understanding of the future implications of OA and climate change is lacking, preliminary management frameworks can be developed to respond to identified future shifts in species' availabilities. To account for the developing understanding of biological responses to OA, management frameworks should be constructed in an adaptable manner to allow for new understandings to be incorporated and for management to respond dynamically. The findings of this thesis analysis, and other related assessments, can be used to guide initial management outlines. However, more detailed future assessments should be considered to fully develop robust management plans.

With the previously discussed potential for international implications of species migrations, a future assessment for the entire East Coast of North America would be an important next step for guiding future management on both sides of the border. The most critical component of this research goal would be a more detailed prediction of redistributions of species. However, including an understanding of the social contexts of dependent communities throughout the range would allow for more equitable allocation of resources under future conditions.

Future assessments of susceptibility to OA and climate change in Atlantic Canada (and along the East Coast of the United States) would benefit from biophysical model outputs at a finer geographic scale. The current iteration of the DBEM uses global-scale climate models for oceanographic conditions, linking the DBEM with higher resolution and/or more localised oceanographic models could allow for a more detailed prediction of potential patterns of change in future fisheries. Similarly, the models informing the DBEM do not account for local amplifiers of OA, which are likely to be highly relevant in Atlantic Canada. This is particularly notable because the Gulf management area, which is semi-enclosed and subject to substantial freshwater input, was projected to be the least exposed management area. If local factors or a regional oceanographic model were used to inform the DBEM it is highly possible that the Gulf management area would have been much more highly affected by OA impacts. An alternative to directly including these local amplifiers into a biophysical model would be for future assessments to consider these local factors as separate contributors to exposure or to use them to weight the biophysical model outputs as done by Ekstrom et al. (2015).

Alternatively, the Atlantis model, with its ecosystem interaction capabilities, might be better suited for regional scale assessments. This model is data intensive and must be adjusted for each ecosystem where it is implemented. However, work with the model has already been commenced in the Northeastern United States (Fay, Link, & Hare, 2017);

expanding this to Atlantic Canada could be a (relatively) easy next step in understanding the effects of OA and climate change across the border.

At the other end of the spatial spectrum, the findings of this thesis could be used to direct future assessments of risk at a sub-provincial scale. In this sense, a future assessment in Atlantic Canada could target the provinces that were found to be at highest risk (i.e., the parts of NB and NS exposed to the Maritime management area, as well as NL and PEI due to their social vulnerability) and assess each of these in more detail in order to identify specific counties/communities which are more and less at risk than the provincial average. In Atlantic provinces rural communities can be much more highly dependent on marine resources than more urban population centres (DFO, 2004; Divovich et al., 2015). Conclusions at this scale would be made more robust with detailed social analyses related to specific employment rates and relationships to other sectors and industries. An interesting, albeit challenging, aspect to include in a provincial scale analysis would be cultural identity associated with fishing. In some respects this is a much more valuable aspect of fisheries employment than any economic indicator. However, as an indicator of sensitivity to changes in fisheries, it would be very difficult to quantify. Qualitative methodology could provide an avenue to incorporate information related to this concept into a framework. Along with finer-scale social data, an analysis at this resolution would require biophysical data to be available at a finer geographic scale. Assessments conducted at a sub-provincial scale would potentially allow for interventions by provincial governments. This level of government is in the best position to deal with many of the factors that contribute to vulnerability related to fisheries. Provincial governments could shift focuses towards alternative industries and develop programs to support transitions for fishery dependent communities.

4.6 Final Thoughts

Climate change, including OA, is going (to continue) to affect human societies on a global scale. Previously stable access to resources will change in response to multiple gradients (IPCC, 2014); fisheries are a prime example of this. Fish stocks are expected to migrate poleward following shifting temperature gradients. However, other climate stressors will also impact future ranges and population stabilities (e.g., Cheung et al., 2010; Portner, 2012). Understanding how human communities rely on and utilise resources such as fisheries will be essential for planning future access and ensuring sustainable use (where possible) in light of shifting distributions (Madin et al., 2012).

Previous research attempting to identify and quantify the social value of fisheries that are potentially threatened by OA has sought to estimate the potential economic costs of anticipated losses in harvest, or identify communities which rely most heavily on susceptible fisheries. To date, most of these assessments have specifically addressed the potential effects of OA in absence of other climate stressors. This research combined direct estimates of changing resource access with an assessment of the social reliance on the resources. Furthermore, this analysis included other climate stressors in attempts to present a more holistic representation of the future fishing conditions in Atlantic Canada.

The more socially vulnerable provinces in the region appear to be somewhat buffered from the stronger negative impacts of changing species distributions. Management decisions and frameworks should actively account for anticipated shifts in previously stable species distributions. This should also be extended to international agreements, as anticipated increases in one nation will inevitably be coming at a cost to another.

There remains a huge amount of uncertainty with respect to the future implications of OA for marine species and ecosystems. To properly address concerns regarding potential human impacts from OA a more thorough understanding of the biophysical impacts is essential – especially with respect to finfish species and interacting climate stressor effects for all species. Addressing these data gaps will require substantial investment in highly controlled experimental studies across a range of species and life-histories. To date much of the experimental research has addressed species which are not directly targeted for harvest. A better understanding of commercial species responses will allow for models of these species to be better informed and generate findings that are more relevant to human systems. Despite the gaps in data, current information on the future of fisheries should be incorporated into management plans to reduce conflicts resulting from otherwise unexpected changes in access to ensure long-term sustainable production from fisheries resources.

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Appendix 1 Fisheries Data

Table A1.1 - DFO landings data for Atlantic Canada for all 'shellfish' species as well as total values for 'groundfish' and 'finfish', values averaged from 1991 to 2010. Annual data available from DFO statistics: http://www.dfo-mpo.gc.ca/stats/commercial/sea-maritimes-eng.htm

	NB		NFL		NS		PEI		Que		Total	
Row Labels	Sum of 20yrAvg Value ('$000) in 2000$	Sum of 20yrAvg Weight (T)	Sum of 20yrAvg Value ('$000) in 2000$	Sum of 20yrAvg Weight (T)	Sum of 20yrAvg Value ('$000) in 2000$	Sum of 20yrAvg Weight (T)	Sum of 20yrAvg Value ('$000) in 2000$	Sum of 20yrAvg Weight (T)	Sum of 20yrAvg Value ('$000) in 2000$	Sum of 20yrAvg Weight (T)	Sum of 20yrAvg Value ('$000) in 2000$	Sum of 20yrAvg Weight (T)
Finfish	144,149	97,316	29,546	68,525	55,005	87,334	8,454	19,436	6,290	9,986	243,444	282,597
Ground-fish	2,676	2,834	59,105	63,302	99,492	95,792	1,856	3,086	10,727	10,043	173,855	175,057
Shellfish	140,994	32,599	325,242	148,159	468,129	133,992	120,556	34,038	119,624	38,825	1,174,545	387,612
Clams	2,301	1,127	17,972	16,368	7,837	7,103	2,872	1,442	1,891	1,575	32,873	27,615
Cockles	-	-	218	682	-	-	-	-	-	-	218	682
Crab, Other	1,439	1,900	439	566	1,843	1,755	1,396	2,290	949	1,305	6,066	7,816
Crab, Queen	46,425	9,577	147,709	45,330	48,521	10,291	7,880	1,696	56,426	13,166	306,961	80,060
Lobster	73,592	8,130	21,078	2,274	272,866	25,070	80,434	9,041	35,522	3,492	483,491	48,008
Mussel	622	474	2,169	1,236	1,380	1,145	16,426	14,296	376	337	20,974	17,487
Other	186	382	805	918	2,252	854	2	3	865	1,018	4,111	3,175
Oyster	2,025	1,048	-	-	948	370	8,899	4,431	-	-	11,871	5,849
Scallop	5,557	3,071	6,416	4,283	85,625	64,745	2,637	834	2,915	1,851	103,151	74,783
Sea cucumber	87	266	63	144	99	302	-	-	21	74	270	786
Sea urchin	2,742	1,237	658	448	1,371	537	8	4	251	190	5,031	2,416
Shrimp	5,993	5,365	126,082	72,670	44,931	21,148	-	-	20,112	15,468	197,119	114,650
Squid	24	22	646	1,996	457	672	0	0	0	0	1,127	2,691
Whelks	-	-	988	1,243	0	0	0	1	294	349	1,282	1,593
Grand Total	287,819	132,748	413,894	279,986	622,626	317,118	130,865	56,560	136,641	58,853	1,591,844	845,266

Appendix 2 DBEM Relative Changes

Table A2.1 - Relative change for each climate RCP scenario and future time step by species for each management area. Values are from median dataset as described in chapter 3 methods, except Gulf snow crab (C. opilio), which is the mean of the climate models' independently aggregated management area values.

Scenario	year	Area	*H. homarus*	*C. opilio*	*P. borealis*	*P. magellanicus*	*M. polynyma*	*M. edulis*	*C. virginica*
RCP 2.6	2050	Gulf	*0.171*	-0.168	*-0.023*	*0.164*	*-0.033*	*0.041*	*0.377*
RCP 2.6	2050	Mar	*-0.120*	*-0.209*	*-0.136*	*-0.071*	*0.040*	*- 0.146*	*0.085*
RCP 2.6	2050	Newf	*0.046*	*-0.128*	*0.100*	*0.043*	*0.064*	*- 0.009*	-
RCP 2.6	2050	tot	*0.024*	*-0.131*	*0.040*	*0.030*	*0.000*	*- 0.025*	*0.179*
RCP 2.6	2090	Gulf	*0.139*	-0.206	*-0.125*	*0.252*	*-0.029*	*0.116*	*0.561*
RCP 2.6	2090	Mar	*-0.185*	*-0.157*	*-0.171*	*-0.127*	*0.028*	*- 0.240*	*0.105*
RCP 2.6	2090	Newf	*0.056*	*-0.161*	*0.097*	*0.065*	*0.091*	*- 0.009*	-
RCP 2.6	2090	tot	*-0.001*	*-0.161*	*0.017*	*0.041*	*0.012*	*- 0.022*	*0.251*
RCP 8.5	2050	Gulf	*0.243*	-0.179	*-0.077*	*0.272*	*-0.219*	*0.154*	*0.212*
RCP 8.5	2050	Mar	*-0.217*	*-0.171*	*-0.198*	*-0.176*	*-0.104*	*- 0.204*	*0.080*
RCP 8.5	2050	Newf	*0.052*	*-0.118*	*0.151*	*0.021*	*0.112*	*0.007*	-
RCP 8.5	2050	tot	*0.013*	*-0.120*	*0.056*	*0.006*	*-0.109*	*0.004*	*0.122*
RCP 8.5	2090	Gulf	*0.269*	-0.390	*-0.292*	*0.471*	*-0.353*	*0.552*	*0.741*
RCP 8.5	2090	Mar	*-0.537*	*-0.328*	*-0.418*	*-0.480*	*-0.433*	*- 0.495*	*0.198*
RCP 8.5	2090	Newf	*0.059*	*-0.168*	*0.297*	*-0.010*	*0.281*	*0.212*	-
RCP 8.5	2090	tot	*-0.080*	*-0.175*	*0.085*	*-0.066*	*-0.150*	*0.156*	*0.370*

Table A2.2 - DBEM projected relative change from individual climate models for each climate scenario and both future time-steps for each species

Climate model	Climate scenario	Region	*C. virginica*	*C. virginica*	*H. homarus*	*H. homarus*	*P. magellanicus*	*P. magellanicus*	*M. edulis*	*M. edulis*	*P. borealis*	*P. borealis*	*C. opilio*	*C. opilio*	*M. polynyma*	*M. polynyma*
			2050	2090	2050	2090	2050	2090	2050	2090	2050	2090	2050	2090	2050	2090
GFDL	RCP 2.6	G	0%	23%	-10%	4%	-1%	14%	-10%	8%	-8%	6%	-100%	-100%	-23%	-17%
IPSL	RCP 2.6	G	44%	68%	28%	15%	20%	28%	37%	43%	11%	-13%	-100%	-100%	2%	-3%
MPI	RCP 2.6	G	38%	65%	14%	15%	19%	27%	6%	14%	-8%	-11%	-17%	-20%	-4%	-5%
GFDL	RCP 8.5	G	-24%	207%	21%	46%	30%	47%	27%	114%	27%	51%	-100%	-100%	-42%	-44%
IPSL	RCP 8.5	G	56%	137%	24%	26%	32%	78%	46%	37%	-7%	-32%	-100%	-100%	-12%	-6%
MPI	RCP 8.5	G	38%	74%	15%	14%	21%	28%	13%	65%	-13%	-31%	-18%	-39%	-12%	-37%
GFDL	RCP 2.6	M	25%	33%	-22%	-31%	-18%	-22%	1%	3%	-18%	-21%	-38%	-47%	-17%	-18%
IPSL	RCP 2.6	M	-1%	2%	-15%	-26%	-5%	-13%	-26%	-44%	-10%	-9%	10%	36%	7%	3%
MPI	RCP 2.6	M	18%	10%	-6%	-11%	0%	-3%	-1%	-5%	-14%	-25%	-15%	-3%	1%	-4%
GFDL	RCP 8.5	M	26%	46%	-33%	-60%	-29%	-57%	4%	16%	-26%	-42%	-40%	-69%	-48%	-48%
IPSL	RCP 8.5	M	4%	-26%	-29%	-65%	-16%	-39%	-44%	-68%	-9%	8%	26%	102%	16%	34%
MPI	RCP 8.5	M	-10%	5%	-11%	-37%	-7%	-39%	0%	-27%	-26%	-61%	-13%	-26%	-24%	-72%
GFDL	RCP 2.6	N	-	-	0%	0%	1%	3%	9%	9%	11%	7%	-17%	-17%	1%	-1%
IPSL	RCP 2.6	N	-	-	10%	13%	10%	11%	-6%	-5%	12%	14%	-9%	-12%	14%	16%
MPI	RCP 2.6	N	-	-	11%	14%	3%	6%	4%	4%	6%	4%	-17%	-13%	4%	3%
GFDL	RCP 8.5	N	-	-	0%	4%	1%	1%	15%	40%	14%	33%	-20%	-7%	3%	12%
IPSL	RCP 8.5	N	-	-	14%	18%	10%	7%	-7%	23%	18%	29%	-9%	-16%	20%	42%
MPI	RCP 8.5	N	-	-	9%	9%	-1%	-6%	8%	13%	14%	31%	-16%	-24%	0%	8%
GFDL	RCP 2.6	T	20%	31%	-7%	-8%	-4%	-2%	3%	8%	5%	3%	-18%	-18%	-12%	-10%
IPSL	RCP 2.6	T	11%	19%	6%	1%	7%	6%	3%	2%	8%	6%	-9%	-10%	7%	4%
MPI	RCP 2.6	T	25%	29%	5%	5%	5%	7%	3%	5%	0%	-4%	-17%	-12%	-1%	-2%
GFDL	RCP 8.5	T	15%	81%	-6%	-7%	-2%	-6%	15%	52%	9%	23%	-21%	-9%	-21%	-18%
IPSL	RCP 8.5	T	17%	17%	2%	-7%	6%	4%	2%	10%	10%	17%	-7%	-11%	1%	13%
MPI	RCP 8.5	T	6%	28%	3%	-6%	1%	-11%	8%	19%	1%	2%	-16%	-25%	-8%	-23%

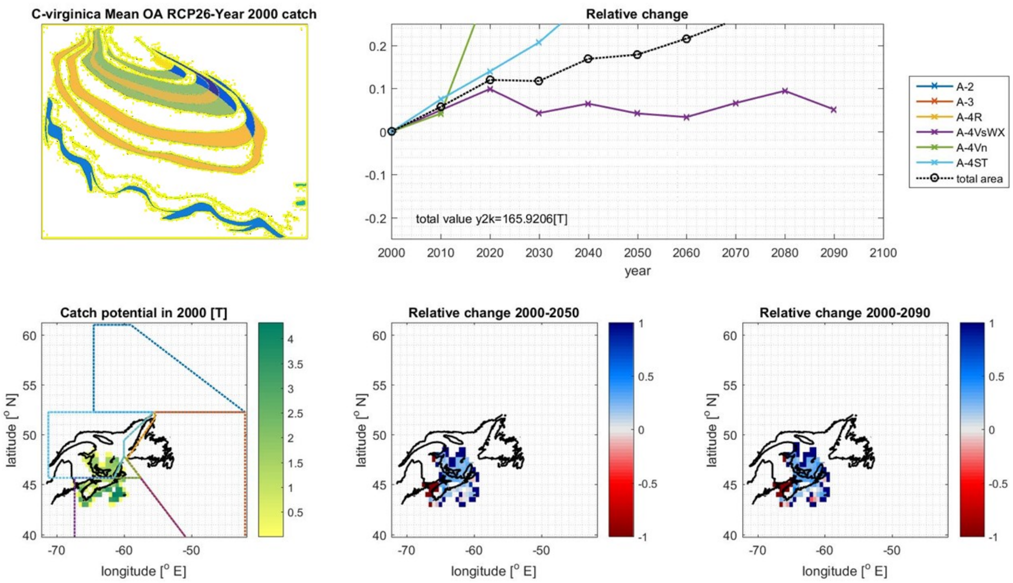
Appendix 3 Relative Distribution Projections

Figure A3.1 - DBEM projections for OA and RCP treatments. Each species is presented in four figures representing: RCP 2.6 'with O,' RCP 2.6 'without OA,' RCP 8.5 'with OA' and RCP 8.5 'without OA,' respectively. Species are presented as follows: a-d = American oyster; e-h American lobster; i-l sea scallop; m-p eastern blue mussel; q-t northern shrimp; u-x snow crab; and y-ab Stimpsons' surf clam.

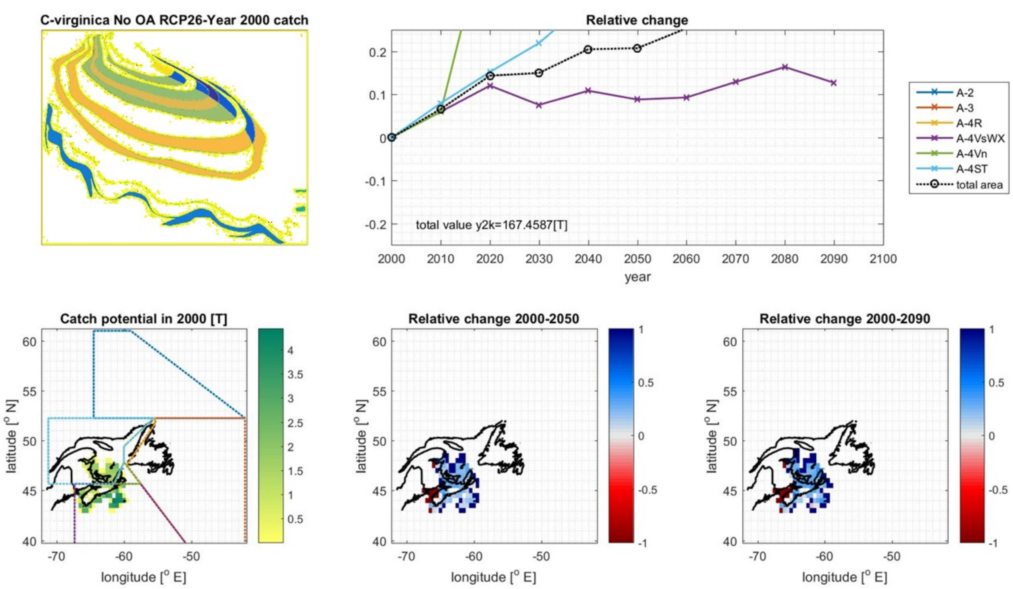
Each figure includes: top left – species graphic; top right – relative change for management areas. Scale is the same for each figure; therefore, species which extend beyond 25% change in either direction go beyond the figure axes (note data separates NAFO subareas that border DFO management areas (i.e. 4R and 4Vn), line colours correspond to distribution map in bottom left). Bottom left – DBEM predicted distribution for baseline year 2000 (averaged 1991-2010). Management borders outline colour corresponds to relative change in top right panel. Bottom middle – relative change in 2050 (average 2041-2060) compared to 2000 (average 1991-2010). Bottom right – relative change for 2090 (average 2081-3000) against 2000 (average 1991-2010).

Note that colour scales presented in bottom-middle and bottom-right panels are relative changes within individual data cell. An artefact of this was that cells which had zero presence in 2000 but gained presence in future time steps had an infinite increase in catch potential; therefore, all increases above 100% were presented as 100%. Relatedly, the vast majority of the cells that experience 100% increases or decreases represent small baseline inputs (as per the bottom right panel).

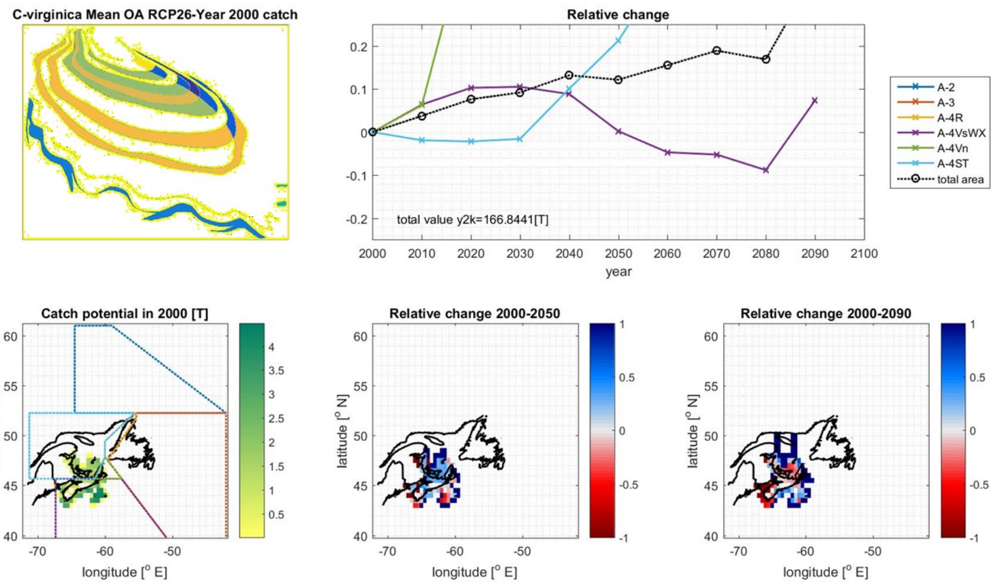
a)


C-virginica Mean OA RCP26-Year 2000 catch
Relative change
total value y2k=165.9206[T]
year
A-2
A-3
A-4R
A-4VsWX
A-4Vn
A-4ST
total area
Catch potential in 2000 [T]
Relative change 2000-2050
Relative change 2000-2090
latitude [° N]
longitude [° E]

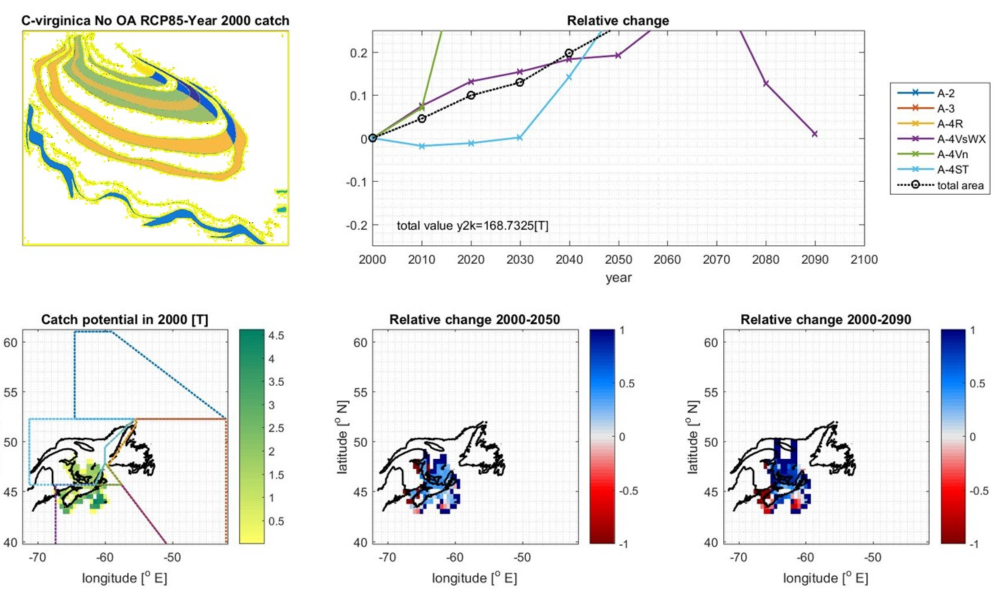
b)


C-virginica No OA RCP26-Year 2000 catch
Relative change
total value y2k=167.4587[T]
year
A-2
A-3
A-4R
A-4VsWX
A-4Vn
A-4ST
total area
Catch potential in 2000 [T]
Relative change 2000-2050
Relative change 2000-2090
latitude [° N]
longitude [° E]

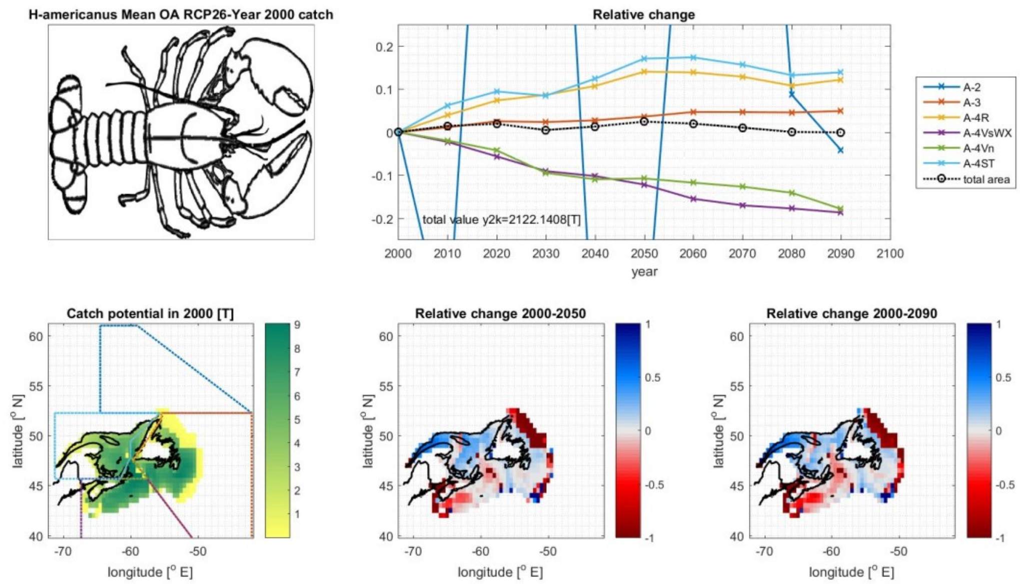
c)


C-virginica Mean OA RCP26-Year 2000 catch
Relative change
total value y2k=166.8441[T]
year
A-2
A-3
A-4R
A-4VsWX
A-4Vn
A-4ST
total area
Catch potential in 2000 [T]
Relative change 2000-2050
Relative change 2000-2090
latitude [° N]
longitude [° E]

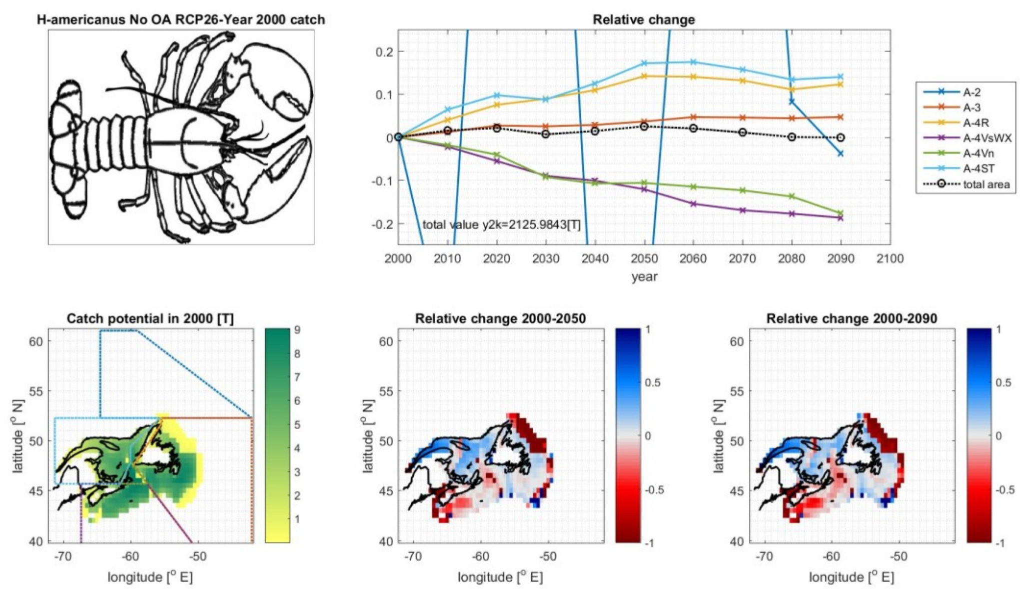
d)


C-virginica No OA RCP85-Year 2000 catch
Relative change
total value y2k=168.7325[T]
year
A-2
A-3
A-4R
A-4VsWX
A-4Vn
A-4ST
total area
Catch potential in 2000 [T]
Relative change 2000-2050
Relative change 2000-2090
latitude [° N]
longitude [° E]

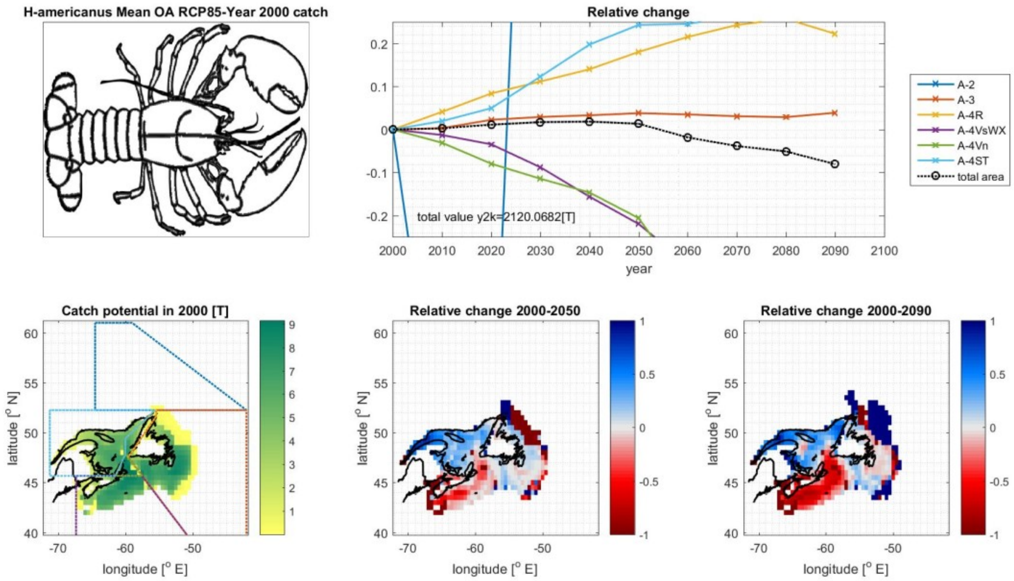
e)


H-americanus Mean OA RCP26-Year 2000 catch
Relative change
A-2
A-3
A-4R
A-4VsWX
A-4Vn
A-4ST
total area
total value y2k=2122.1408[T]
year
Catch potential in 2000 [T]
Relative change 2000-2050
Relative change 2000-2090
latitude [° N]
longitude [° E]

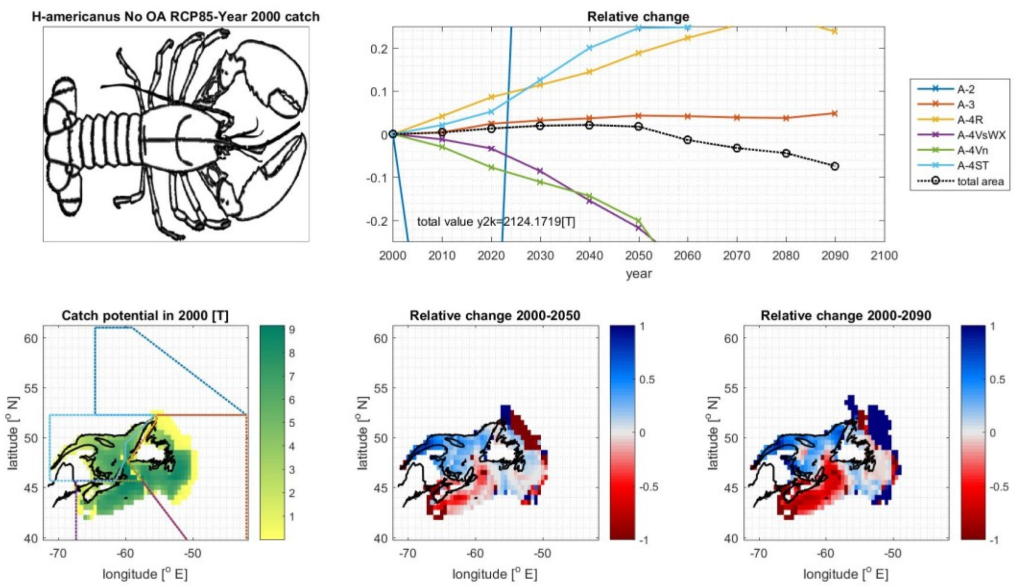
f)


H-americanus No OA RCP26-Year 2000 catch
Relative change
A-2
A-3
A-4R
A-4VsWX
A-4Vn
A-4ST
total area
total value y2k=2125.9843[T]
year
Catch potential in 2000 [T]
Relative change 2000-2050
Relative change 2000-2090
latitude [° N]
longitude [° E]

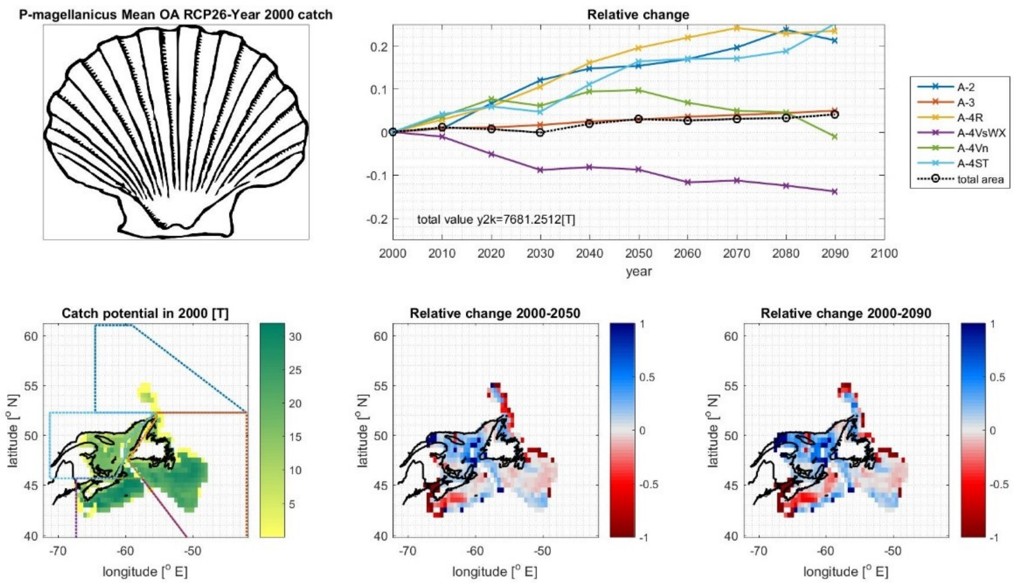
g)


H-americanus Mean OA RCP85-Year 2000 catch
Relative change
total value y2k=2120.0682[T]
year
A-2
A-3
A-4R
A-4VsWX
A-4Vn
A-4ST
total area
Catch potential in 2000 [T]
Relative change 2000-2050
Relative change 2000-2090
latitude [° N]
longitude [° E]

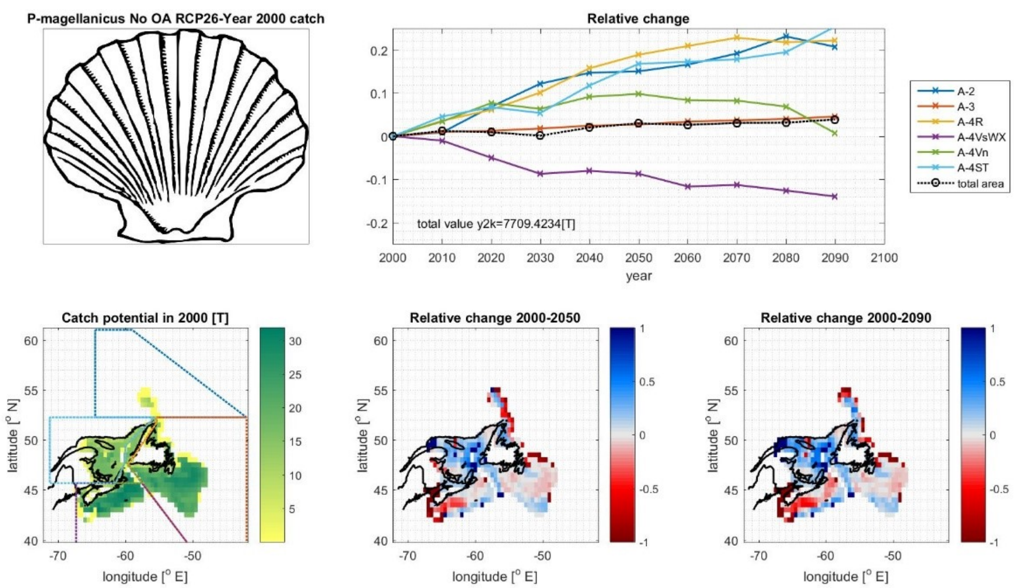
h)


H-americanus No OA RCP85-Year 2000 catch
Relative change
total value y2k=2124.1719[T]
year
A-2
A-3
A-4R
A-4VsWX
A-4Vn
A-4ST
total area
Catch potential in 2000 [T]
Relative change 2000-2050
Relative change 2000-2090
latitude [° N]
longitude [° E]

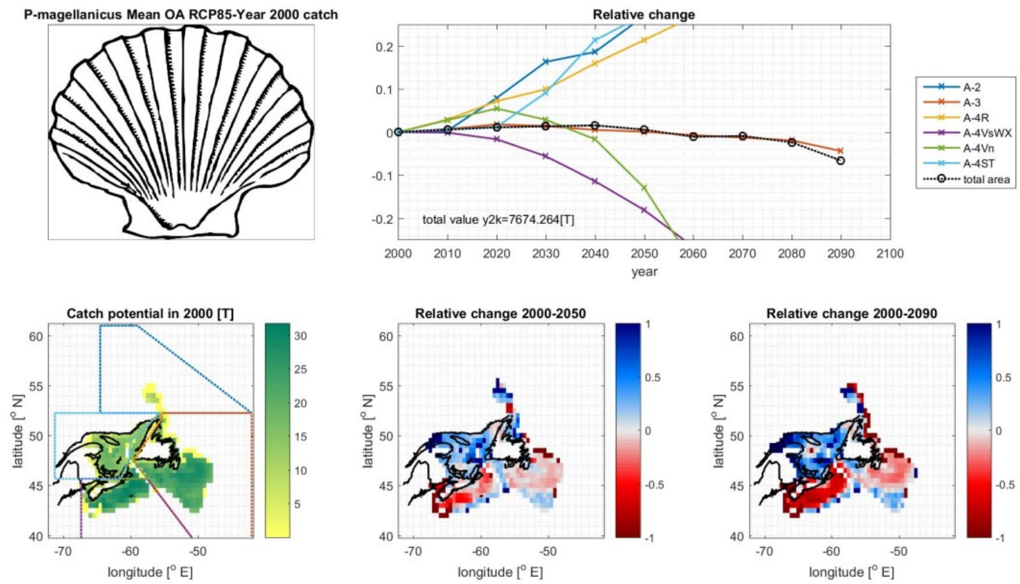
i)


P-magellanicus Mean OA RCP26-Year 2000 catch
Relative change
A-2
A-3
A-4R
A-4VsWX
A-4Vn
A-4ST
total area
total value y2k=7681.2512[T]
year
Catch potential in 2000 [T]
Relative change 2000-2050
Relative change 2000-2090
latitude [° N]
longitude [° E]

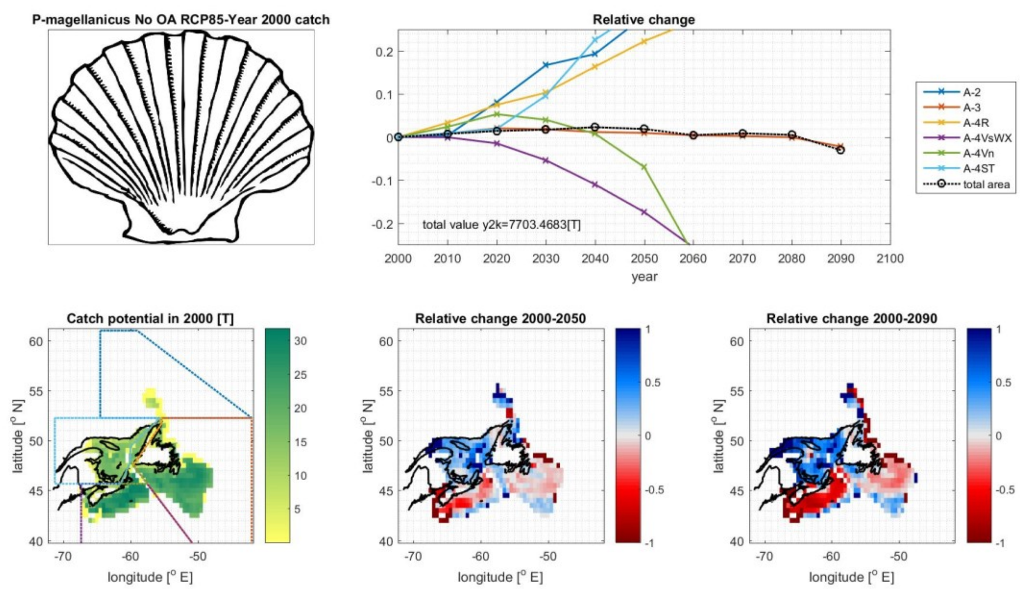
j)


P-magellanicus No OA RCP26-Year 2000 catch
Relative change
A-2
A-3
A-4R
A-4VsWX
A-4Vn
A-4ST
total area
total value y2k=7709.4234[T]
year
Catch potential in 2000 [T]
Relative change 2000-2050
Relative change 2000-2090
latitude [° N]
longitude [° E]

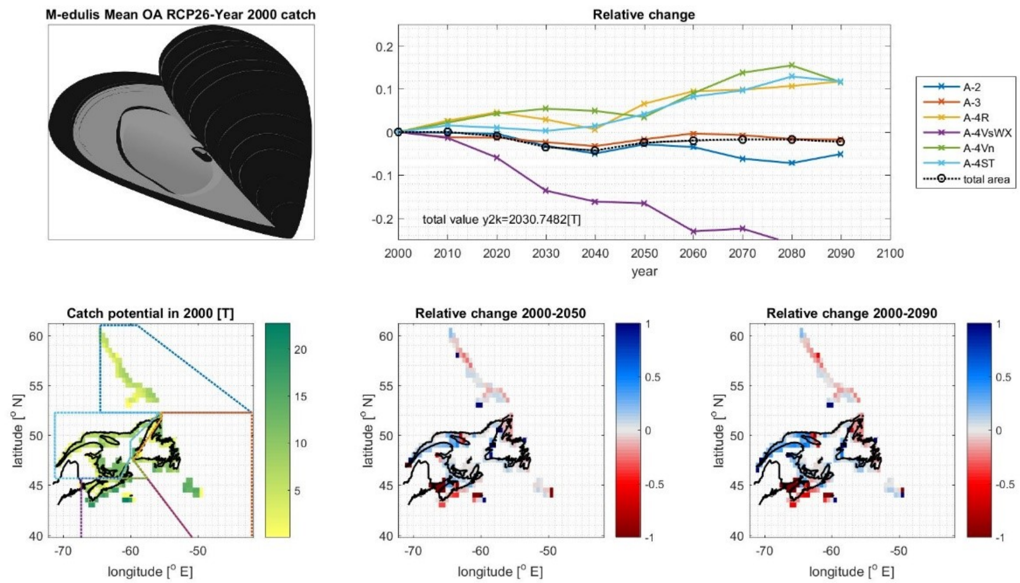
k)


P-magellanicus Mean OA RCP85-Year 2000 catch
Relative change
A-2
A-3
A-4R
A-4VsWX
A-4Vn
A-4ST
total area
total value y2k=7674.264[T]
year
Catch potential in 2000 [T]
Relative change 2000-2050
Relative change 2000-2090
latitude [° N]
longitude [° E]

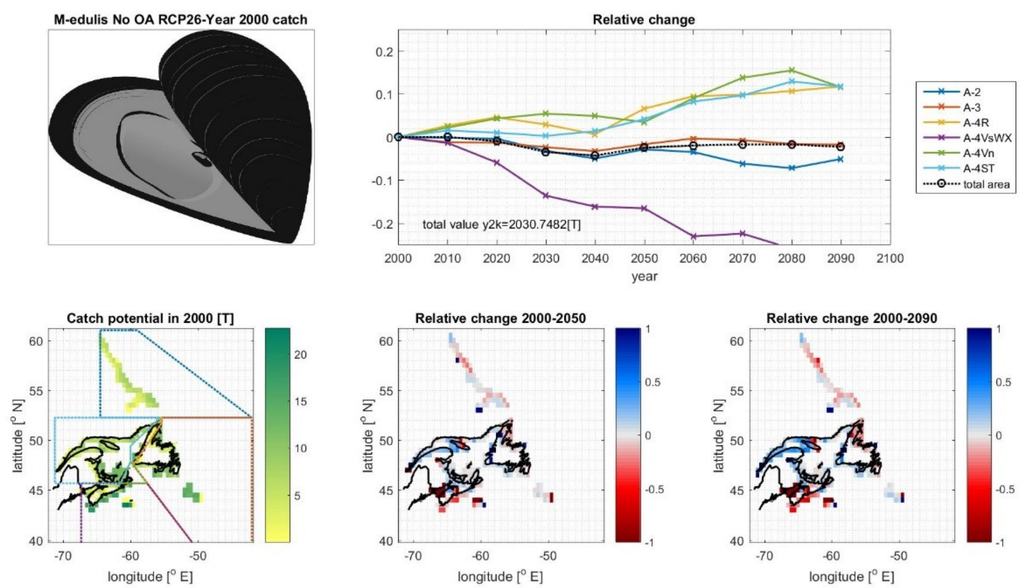
l)


P-magellanicus No OA RCP85-Year 2000 catch
Relative change
A-2
A-3
A-4R
A-4VsWX
A-4Vn
A-4ST
total area
total value y2k=7703.4683[T]
year
Catch potential in 2000 [T]
Relative change 2000-2050
Relative change 2000-2090
latitude [° N]
longitude [° E]

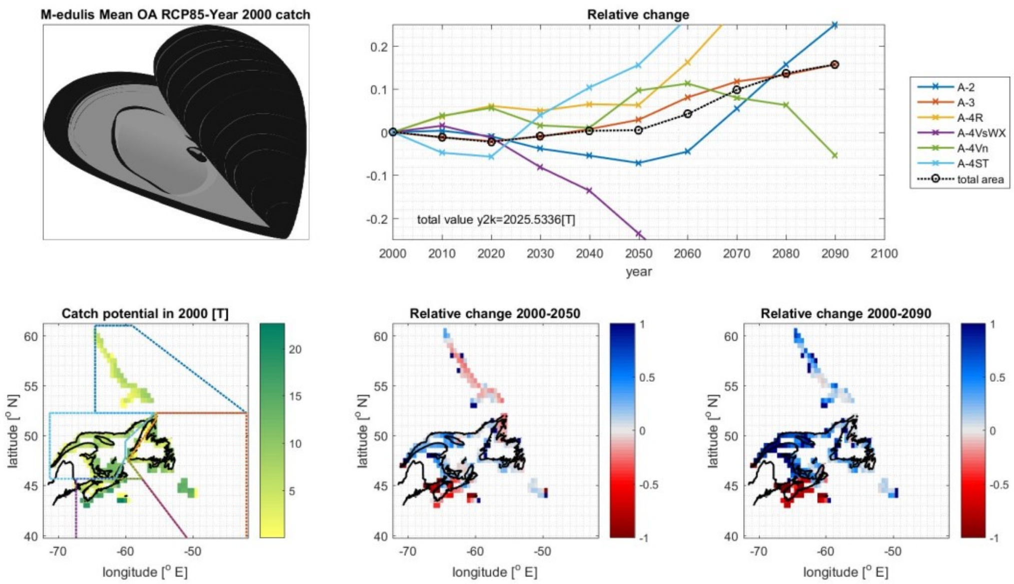
m)


M-edulis Mean OA RCP26-Year 2000 catch
Relative change
A-2
A-3
A-4R
A-4VsWX
A-4Vn
A-4ST
total area
total value y2k=2030.7482[T]
year
Catch potential in 2000 [T]
Relative change 2000-2050
Relative change 2000-2090
latitude [° N]
longitude [° E]

n)

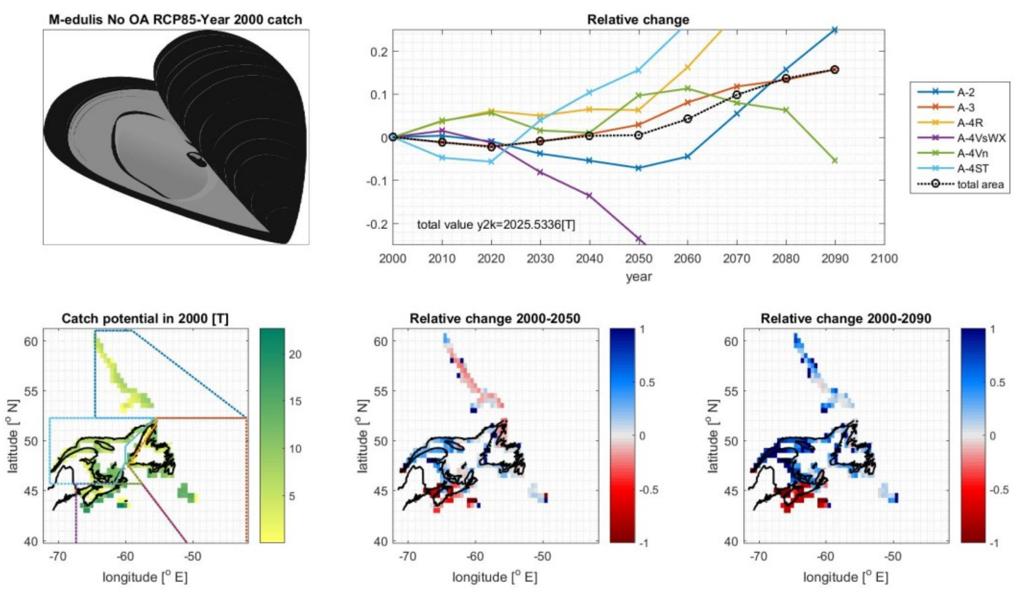

M-edulis No OA RCP26-Year 2000 catch
Relative change
A-2
A-3
A-4R
A-4VsWX
A-4Vn
A-4ST
total area
total value y2k=2030.7482[T]
year
Catch potential in 2000 [T]
Relative change 2000-2050
Relative change 2000-2090
latitude [° N]
longitude [° E]

o)



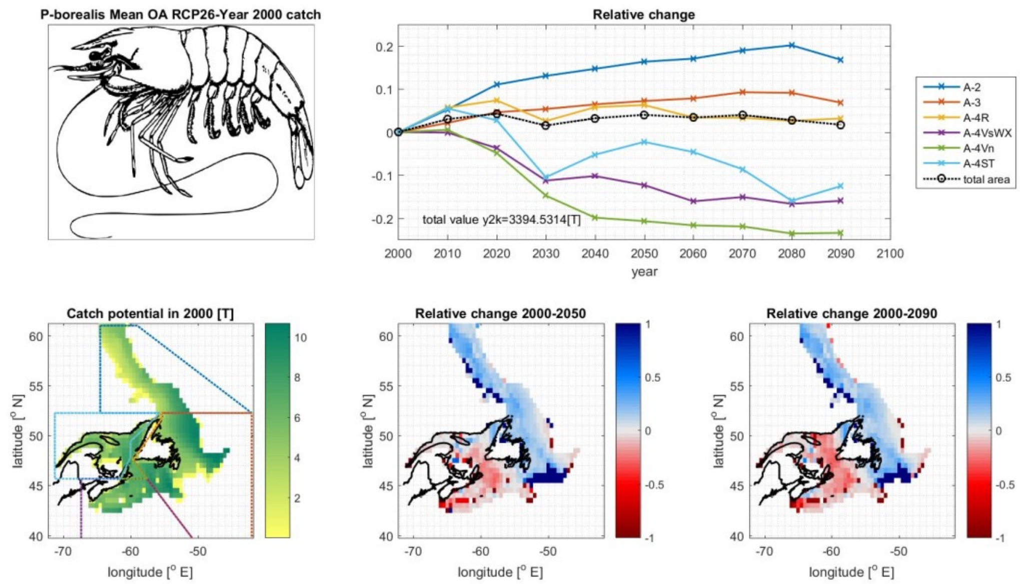
M-edulis Mean OA RCP85-Year 2000 catch
Relative change
A-2
A-3
A-4R
A-4VsWX
A-4Vn
A-4ST
total area
total value y2k=2025.5336[T]
year
Catch potential in 2000 [T]
Relative change 2000-2050
Relative change 2000-2090
latitude [° N]
longitude [° E]

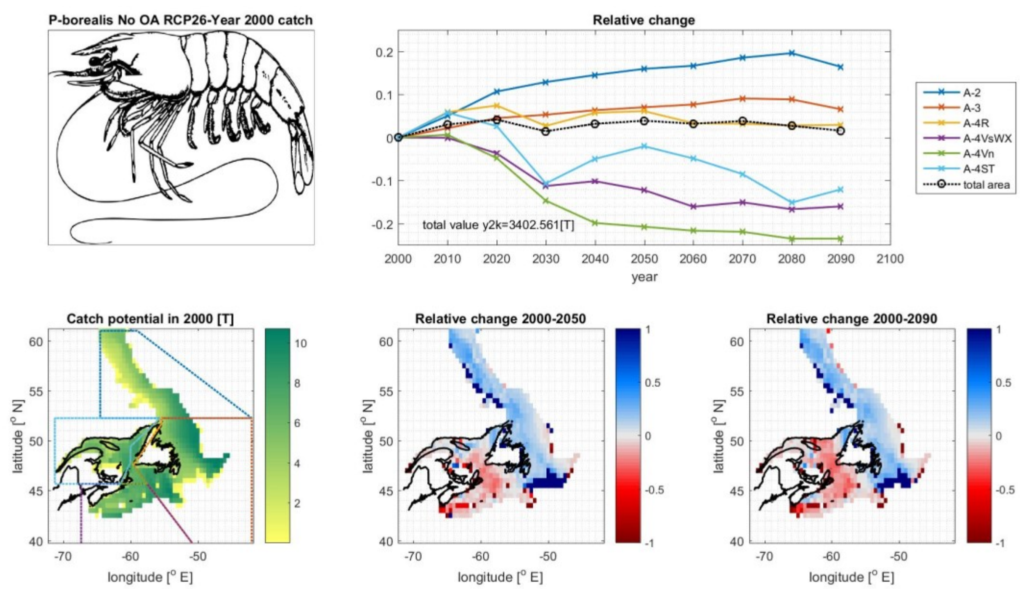
p)



M-edulis No OA RCP85-Year 2000 catch
Relative change
A-2
A-3
A-4R
A-4VsWX
A-4Vn
A-4ST
total area
total value y2k=2025.5336[T]
year
Catch potential in 2000 [T]
Relative change 2000-2050
Relative change 2000-2090
latitude [° N]
longitude [° E]

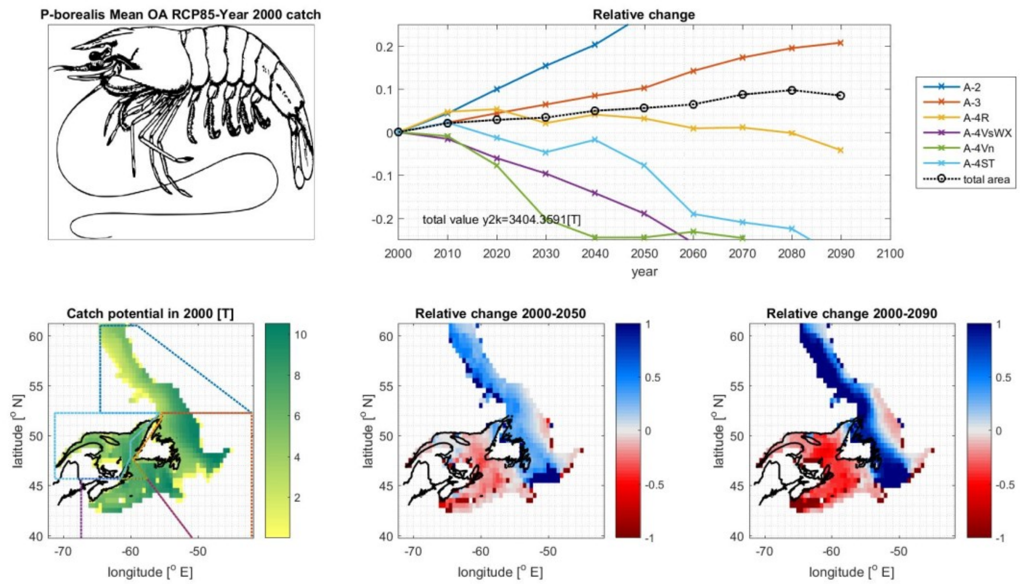
q)


P-borealis Mean OA RCP26-Year 2000 catch
Relative change
total value y2k=3394.5314[T]
year
A-2
A-3
A-4R
A-4VsWX
A-4Vn
A-4ST
total area
Catch potential in 2000 [T]
Relative change 2000-2050
Relative change 2000-2090
latitude [° N]
longitude [° E]

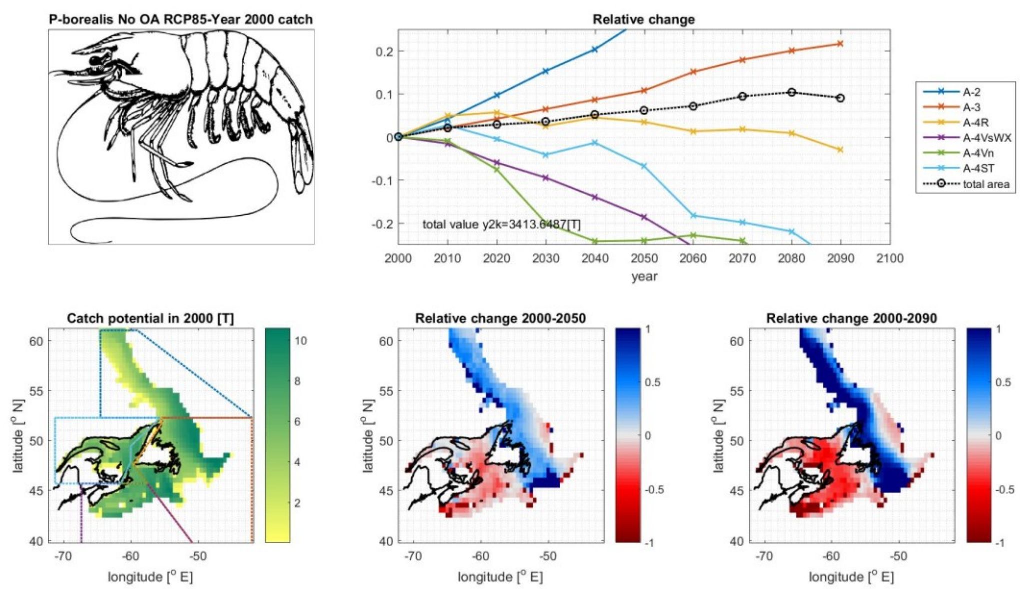

P-borealis No OA RCP26-Year 2000 catch
Relative change
total value y2k=3402.561[T]
year
A-2
A-3
A-4R
A-4VsWX
A-4Vn
A-4ST
total area
Catch potential in 2000 [T]
Relative change 2000-2050
Relative change 2000-2090
latitude [° N]
longitude [° E]

r)

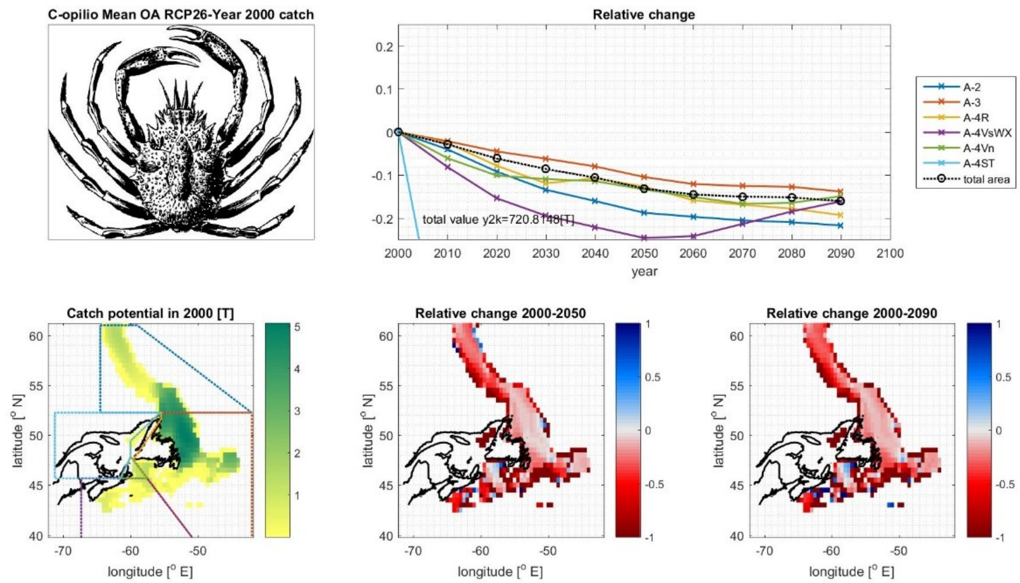
s)


P-borealis Mean OA RCP85-Year 2000 catch
Relative change
total value y2k=3404.3591[T]
year
A-2
A-3
A-4R
A-4VsWX
A-4Vn
A-4ST
total area
Catch potential in 2000 [T]
Relative change 2000-2050
Relative change 2000-2090
latitude [° N]
longitude [° E]

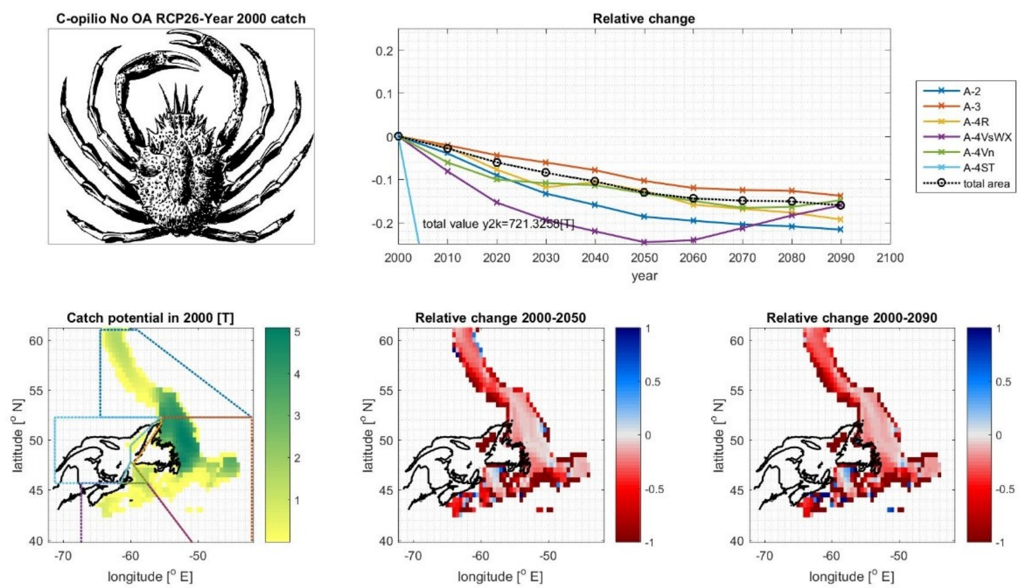
t)


P-borealis No OA RCP85-Year 2000 catch
Relative change
total value y2k=3413.6487[T]
year
A-2
A-3
A-4R
A-4VsWX
A-4Vn
A-4ST
total area
Catch potential in 2000 [T]
Relative change 2000-2050
Relative change 2000-2090
latitude [° N]
longitude [° E]

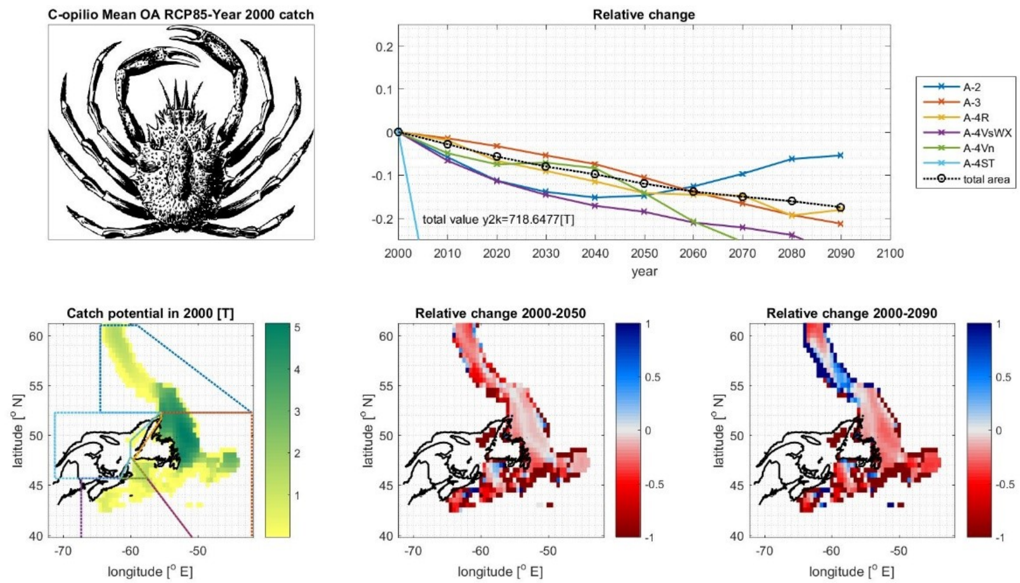
u)


C-opilio Mean OA RCP26-Year 2000 catch
Relative change
A-2
A-3
A-4R
A-4VsWX
A-4Vn
A-4ST
total area
total value y2k=720.8148[T]
year
Catch potential in 2000 [T]
Relative change 2000-2050
Relative change 2000-2090
latitude [° N]
longitude [° E]

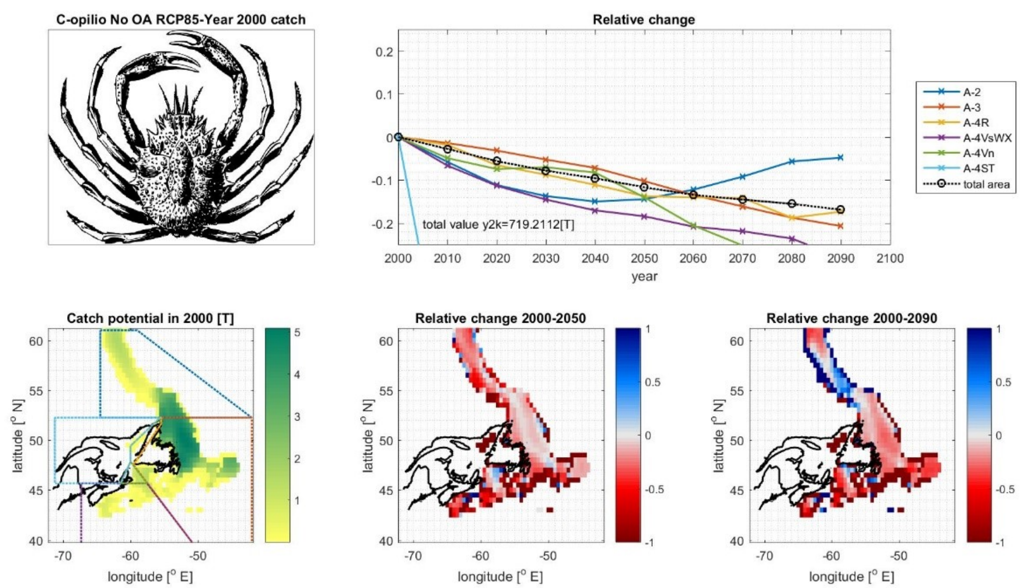
v)


C-opilio No OA RCP26-Year 2000 catch
Relative change
A-2
A-3
A-4R
A-4VsWX
A-4Vn
A-4ST
total area
total value y2k=721.3258[T]
year
Catch potential in 2000 [T]
Relative change 2000-2050
Relative change 2000-2090
latitude [° N]
longitude [° E]

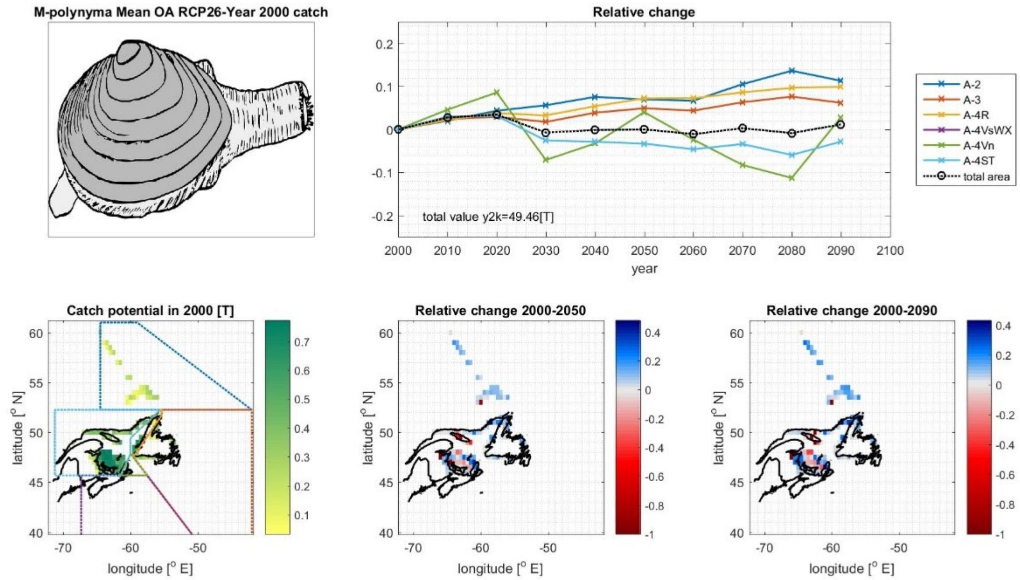
w)


C-opilio Mean OA RCP85-Year 2000 catch
Relative change
A-2
A-3
A-4R
A-4VsWX
A-4Vn
A-4ST
total area
total value y2k=718.6477[T]
year
Catch potential in 2000 [T]
Relative change 2000-2050
Relative change 2000-2090
latitude [° N]
longitude [° E]

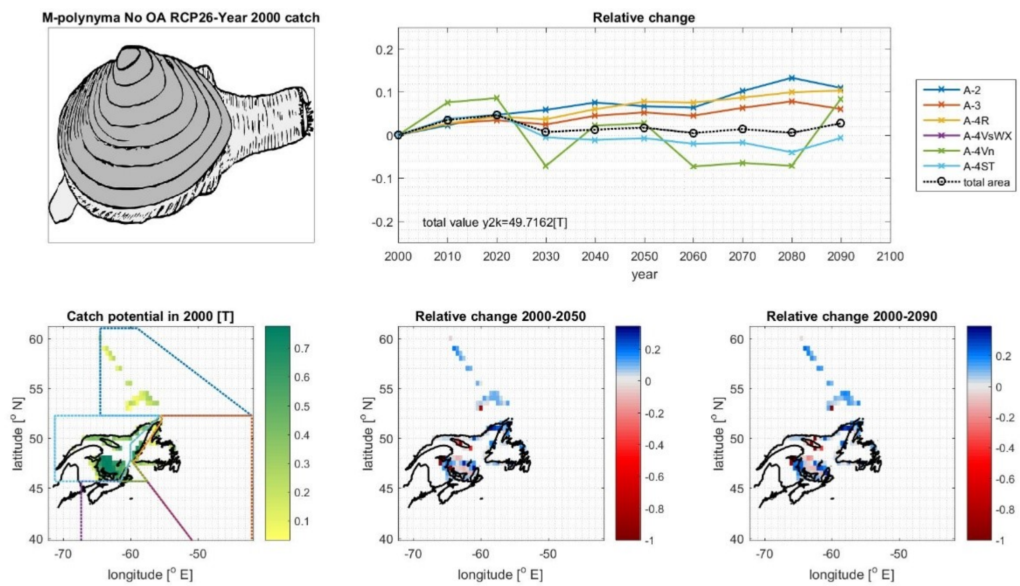
x)


C-opilio No OA RCP85-Year 2000 catch
Relative change
A-2
A-3
A-4R
A-4VsWX
A-4Vn
A-4ST
total area
total value y2k=719.2112[T]
year
Catch potential in 2000 [T]
Relative change 2000-2050
Relative change 2000-2090
latitude [° N]
longitude [° E]

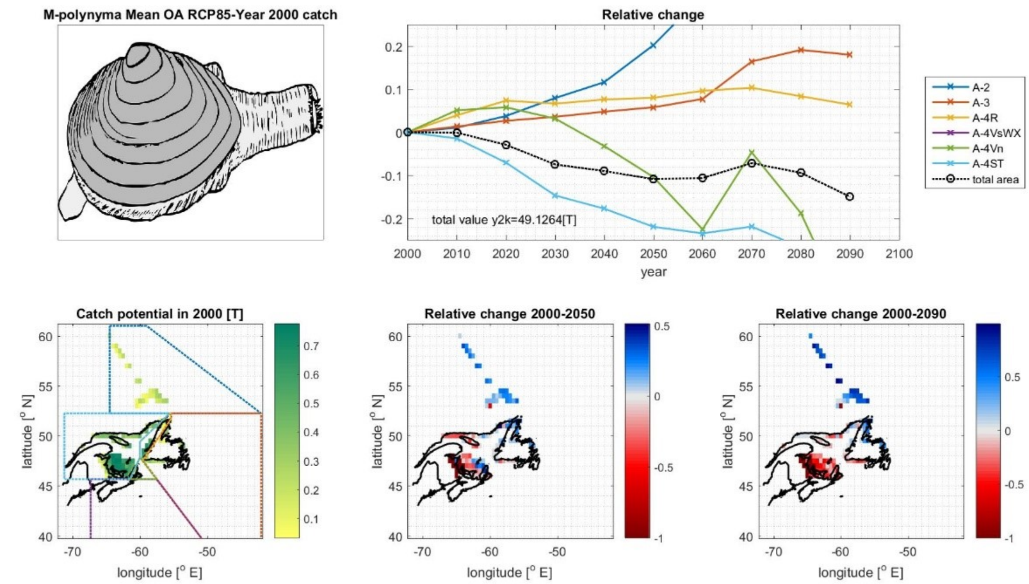
y)


M-polynyma Mean OA RCP26-Year 2000 catch
Relative change
total value y2k=49.46[T]
year
A-2
A-3
A-4R
A-4VsWX
A-4Vn
A-4ST
total area
Catch potential in 2000 [T]
Relative change 2000-2050
Relative change 2000-2090
latitude [° N]
longitude [° E]

z)

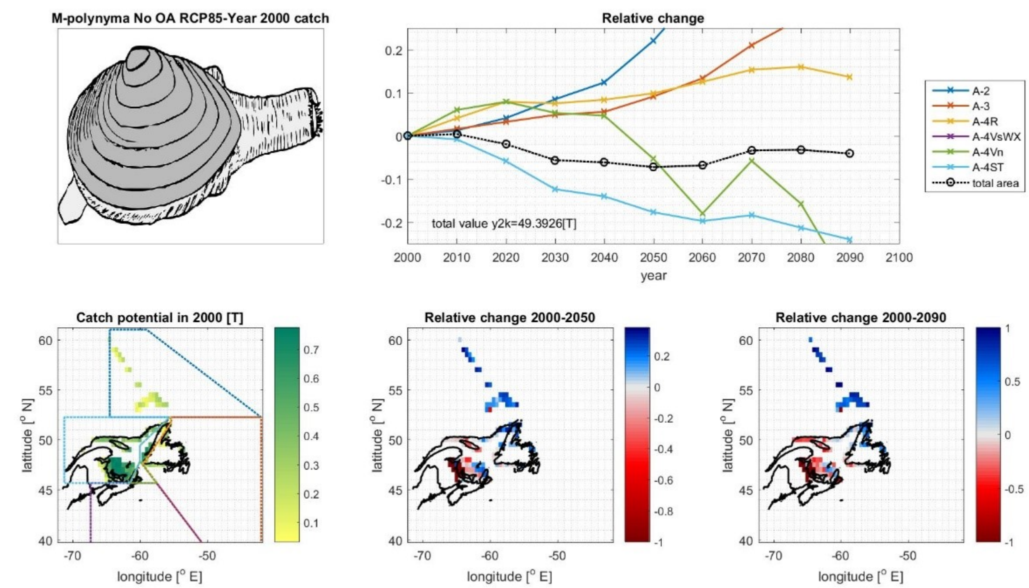

M-polynyma No OA RCP26-Year 2000 catch
Relative change
total value y2k=49.7162[T]
year
A-2
A-3
A-4R
A-4VsWX
A-4Vn
A-4ST
total area
Catch potential in 2000 [T]
Relative change 2000-2050
Relative change 2000-2090
latitude [° N]
longitude [° E]

aa)



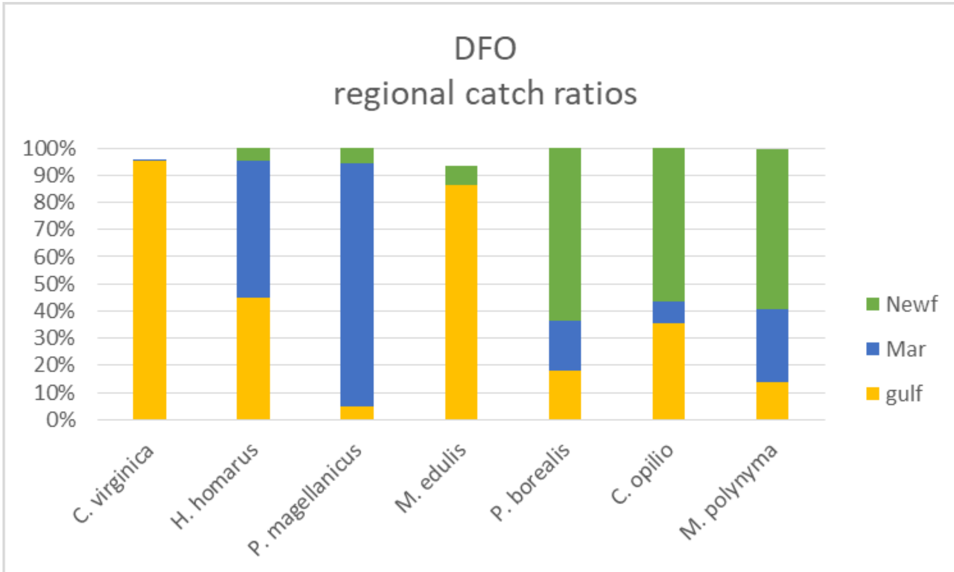
M-polynyma Mean OA RCP85-Year 2000 catch
Relative change
A-2
A-3
A-4R
A-4VsWX
A-4Vn
A-4ST
total area
total value y2k=49.1264[T]
year
Catch potential in 2000 [T]
Relative change 2000-2050
Relative change 2000-2090
latitude [° N]
longitude [° E]

ab)



M-polynyma No OA RCP85-Year 2000 catch
Relative change
A-2
A-3
A-4R
A-4VsWX
A-4Vn
A-4ST
total area
total value y2k=49.3926[T]
year
Catch potential in 2000 [T]
Relative change 2000-2050
Relative change 2000-2090
latitude [° N]
longitude [° E]

Appendix 4 DBEM and DFO Management Area Landings for 2000



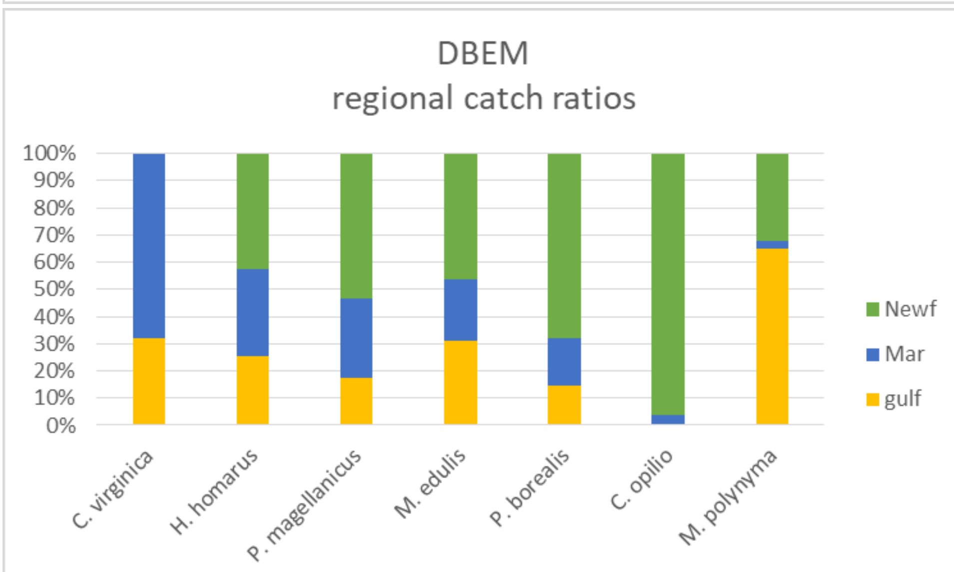


Figure A5.1 - DFO (top) and DBEM (bottom) distribution species catches across management areas. Note that DFO data are compared against total reported catch for each species and therefore may not sum to 100%.

Appendix 5 Sensitivity Testing

Table A4.1 - Scores for individual risk components used for sensitivity testing in subsequent table.

Province	Mgmt. area	RCP scenario	Exposure 2050	Exposure 2090	Sensitivity	Adaptive capacity	Vulnerability
NB	Gulf	2.6	0.049	0.092	0.193	0.749	0.471
		8.5	0.000	0.063			
	Mar	2.6	0.324	0.416			
		8.5	0.479	1.000			
NL	Newf	2.6	0.146	0.169	0.617	0.897	0.757
		8.5	0.097	0.019			
NS	Gulf	2.6	0.049	0.092	0.432	0.037	0.234
		8.5	0.000	0.063			
	Mar	2.6	0.324	0.416			
		8.5	0.479	1.000			
PEI	Gulf	2.6	0.049	0.092	1.000	0.467	0.734
		8.5	0.000	0.063			
Que	Gulf	2.6	0.049	0.092	0.000	0.272	0.136
		8.5	0.000	0.063			

Figure A4.1 – Alternate framework orientations for sensitivity testing. Letters correspond to sensitivity testing risk outcomes in Table A4.2

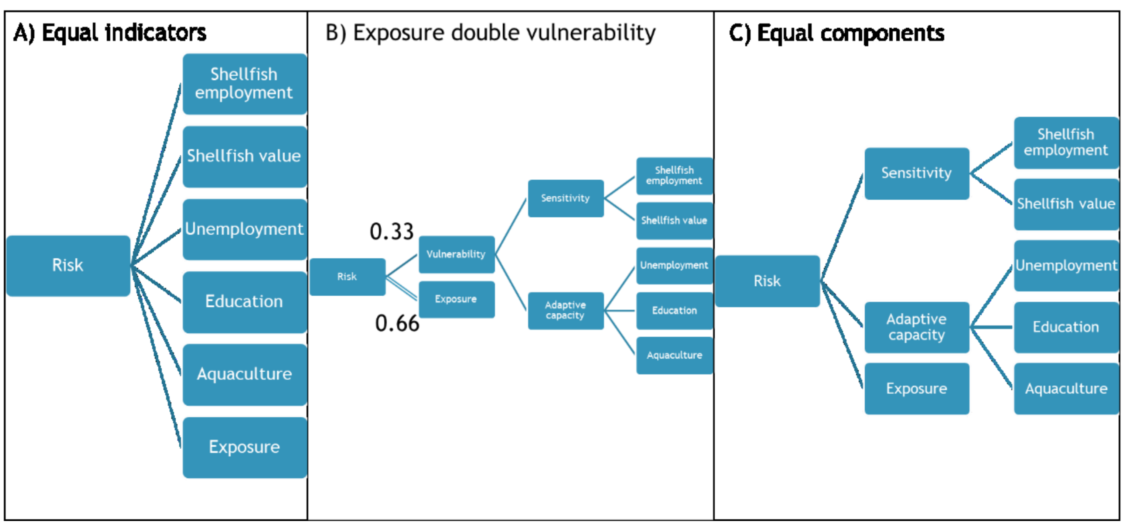


Table A4.2 - Risk index results for 3 alternative weighting schemes. Top six highest risk scores in bold for each treatment. Framework orientations schematics appear in Figure A4.1

Province	Mgmt. area	RCP scenario	A) Equal weighting of individual indicators		B) Exposure weighted 2, vulnerability weighted 1.		C) Exposure = Sensitivity = Adaptive capacity	
			2050	2090	2050	2090	2050	2090
NB	Gulf	2.6	0.45	0.45	0.19	0.22	0.33	0.35
		8.5	0.44	0.45	0.16	0.20	0.31	0.34
	Mar	2.6	0.49	0.51	**0.37**	**0.43**	0.42	0.45
		8.5	0.52	**0.61**	**0.48**	**0.82**	0.47	**0.65**
NL	Newf	2.6	**0.68**	**0.68**	0.35	0.37	**0.55**	**0.56**
		8.5	**0.67**	**0.66**	0.32	0.26	**0.54**	**0.51**
NS	Gulf	2.6	0.17	0.18	0.11	0.14	0.17	0.19
		8.5	0.16	0.17	0.08	0.12	0.16	0.18
	Mar	2.6	0.22	0.23	0.29	0.36	0.26	0.30
		8.5	0.24	0.33	**0.40**	**0.74**	0.32	0.49
PEI	Gulf	2.6	0.58	**0.58**	0.28	0.31	0.51	**0.52**
		8.5	0.57	0.58	0.24	0.29	0.49	0.51
Que	Gulf	2.6	0.14	0.15	0.08	0.11	0.11	0.12
		8.5	0.14	0.15	0.05	0.09	0.09	0.11