MARINE CLIMATE CHANGE VULNERABILITY ASSESSMENT DEVELOPMENT, USES, AND LIMITATIONS AS A TOOL FOR CLIMATE ADAPTATION

by

Christine H. Stortini

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DEDICATION PAGE

This thesis is dedicated to my best friend and husband-to-be, Jordan, to my beautiful mother, and to my loving, ever-encouraging big brother, Ben.

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ABSTRACT

Resource managers need tools to prepare for biological changes resulting from climate change. Here, I present a novel marine Vulnerability to Projected Warming Assessment (VPWA) model as a climate adaptation tool to rank local species by the risk imposed to their future distribution and abundance by projected regional warming. This VPWA builds on earlier models; it includes a species distribution model and thermal window analyses to project changes in regionally available thermal habitat at multiple life stages, provides refined vulnerability ranks, and weights model factors by importance. I assess 33 fish and invertebrate species on the Scotian Shelf under two warming scenarios. At smaller spatial scales, I evaluate populations of seven of these species. Identification of species/populations with high vulnerability scores may help managers prioritize resources and identify knowledge gaps. This tool can and should be improved to include pH, oxygen, stratification, and ecosystem-related concerns as more data become available.

LIST OF ABBREVIATIONS USED

ACCASP	Aquatic Climate Change Adaptation Services Program
CCVA	Climate Change Vulnerability Assessment
DFO	Fisheries and Oceans Canada (Department of Fisheries and Oceans)
ESS	Eastern Scotian Shelf
IPCC	Intergovernmental Panel on Climate Change
SDM	Species Distribution Model
SS	Scotian Shelf, Canada
WSS	Western Scotian Shelf
VA	Vulnerability Assessment
VPWA	Vulnerability to Projected Warming Assessment

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CHAPTER 1 INTRODUCTION

Climate change in the ocean poses risks to marine species/populations and therefore to human communities and industries. Marine managers face the challenge of managing under uncertainty; living marine resources will shift in abundance, existence, and distribution as a result of climate change, but how? Consequently, this thesis work was initiated to improve upon an existing tool (the Climate Change Vulnerability Assessment (CCVA)) to help marine managers predict and prepare for climate-induced changes in ocean biology. Currently available CCVAs can be improved to increase relevance at regional scales, the scale at which marine managers plan. This thesis aims to build on currently available CCVA models to provide refined ranks for species vulnerability and to include three-dimensional spatial anayses, so that the applicability of results to regional marine management programs will be improved.

1.1 CLIMATE CHANGE

Our planet is experiencing climatic changes as a result of the accumulation of greenhouse gases in the atmosphere since the industrial revolution. Long-term climate trends show significant deviation from natural variability, and warming exceeds trends from our geological past (IPCC, 2013). As the ocean drives all climate and weather patterns on the planet, it has suffered the brunt of the climate change problem, storing approximately 90% of the total heat trapped by the greenhouse effect since 1971 (IPCC, 2013). Warming is especially evident near the poles where it is causing a positive feedback loop, accelerating the melting of land and sea ice, reducing albedo and accelerating warming. Indeed, the extent of Arctic sea ice has retreated by approximately

3.5-4.1% per decade since 1979, and is projected to melt another 43-94% by the year 2100 (IPCC, 2013). Increased fresh water into the ocean via melting ice can cause increased stratification, because fresh water "floats" atop the saline water, creating a density divide. This can reduce mixing, which can reduce the amount of oxygen and food transported to deep-water habitats (Kennett and Stott, 1991; Ally, 2007). Further, oxygen dissolves less readily in warm water than it does in cold water; therefore oxygen levels are projected to decrease as warming continues (Blumberg and Di Toro, 1990; Gilbert *et al.*, 2005).

Warming, and its related impacts, is only one side effect of increased greenhouse gases. The ocean has absorbed approximately 30% of the CO_2 that has been emitted to the atmosphere (IPCC, 2013). This CO_2 reacts with water in a way that reduces concentrations of carbonate compounds and increases the concentration of hydrogen ions (H+), making the water more acidic (Sarmiento and Gruber, 2006). This phenomenon is called "ocean acidification". Reductions in carbonate concentrations can impact many organisms that require carbonate to build their shells (Orr *et al.*, 2005; Gruber, 2011). In extremely acidic conditions, carbonate shells can dissolve. Since the beginning of the industrial era, there has been a decrease in average global ocean pH of 0.1 units (IPCC, 2013). It is projected that, by the year 2100, ocean acidity could reach levels that have been implicated in past mass extinctions (Orr *et al.*, 2005).

On a global scale, climate change presents many issues for the human population. Melting of sea ice is not only accelerating ocean warming and increasing stratification, it is causing sea level to rise. Global mean sea level has risen by 0.17 to 0.21m over the last century (IPCC, 2013). It is suggested that sea level rise may accelerate, and consequently produce millions of "environmental refugees" (Myers, 2002). Further, warming, lowered oxygen, increased stratification, and acidification are likely to, if they have not already, impact marine organisms and their communities. Marine fish and invertebrates typically reside within a range of tolerable temperatures; therefore warming can force migration and/or have negative physiological impacts (Neuheimer *et al*., 2011). Reduced oxygen can have equally, if not more severe impacts on marine animals as they often depend on oxygen for survival (Gilly *et al*., 2013). Further decreased oxygen can impact microbial cycles. Melting ice will likely impact polar animals which depend on ice as feeding and/or breeding habitat (Johannessen and Miles, 2010; Post *et al*., 2013). Stratification can limit the transport of oxygen and nutrients to benthic habitats (Ally, 2007), and acidification may have significant ecosystem-level impacts (Gruber, 2011). Negative impacts on marine animals will consequently impact human communities, which depend on living marine resources for food and ecosystem services (Shackell *et al*., 2013).

The Intergovernmental Panel on Climate Change (IPCC), an international body of scientists dedicated to the assessment and projection of climate change and its impacts, discovered that observed climate changes are heavily correlated to anthropogenic greenhouse gas emissions since the industrial revolution (IPCC, 2013). The largest contributor to climate change has been the increase in the atmospheric concentration of CO_2. In fact, emissions of CO_2 have increased by 40% since pre-industrial times (IPCC, 2013). In 1997, countries bound themselves to the Kyoto Protocol in an effort to slow climate change (UNFCCC, 2014). This meant they were obligated to reduce greenhouse gas emissions by at least 5% below 1990 levels by the year 2012. For many countries, this goal was not met. Fewer countries have now committed to reducing greenhouse gas emissions by at least 18% below 1990 levels by the year 2020 (UNFCCC, 2014). However, even if all countries stopped emitting greenhouse gases immediately, climate change would continue for several centuries (IPCC, 2013); the trajectory has been set in place and changes are accelerating. While human society as a whole continues to discuss and negotiate how we are to slow or stop climate change, governments are recognizing the need to adapt for the changes that are now unavoidable.

1.1.1 National problems

Canada is unique in that it has more ocean coastline than any other country (NRC, 2004). It measures more than 240 000 kilometers in length. Further, the country's ocean estate is approximately 7.1 million km^2, which corresponds to 70% of the country's landmass (DFO, 2014b). The livelihood of a large portion of the country's population depends on industries related to the ocean, mainly oil and gas, shipping and transportation, tourism, and fishing. Canada's ocean industries currently contribute $26 billion per year to the national GDP, and employ upwards of 315 000 Canadians (DFO, 2013a).

Because ocean communities and industries represent a crucial part of Canada's culture and economy, climate change is especially threatening here. With rising sea level and increased intensity and frequency of storms, coastal communities could be at risk of land loss and structural damage (Shackell *et al.*, 2013). Additionally, the risk associated with ocean-based activities may increase. Chemical and physical changes in the ocean could also cause shifts in the availability of biological resources (e.g., Cheung *et al.*, 2011; Shackell *et al.*, 2013). In order to avoid economic damage from unavoidable climate-induced changes, the Canadian government has recognized that adaptation programs must be put into place within many marine management divisions (DFO, 2014a).

1.1.2 Regional management – Governing under uncertainty

In Canada, there are many different “bioregions” within the three oceans - Atlantic, Arctic, and Pacific - lining our coast (Government of Canada, 2011). Regional offices or ‘divisions’ of Fisheries and Oceans Canada manage these “bioregions”. Included in these bioregions are the Scotian Shelf and slope, Gulf of St Lawrence and St Lawrence estuary, Newfoundland-Labrador shelves, eastern Arctic, Arctic Archipelago, Hudson Bay, Arctic basin, western Arctic, Strait of Georgia, Northern Shelf, Southern Shelf, and Offshore Pacific (Government of Canada, 2011). Each bioregion and the living resources within them will be affected by climate change differently. It is important to understand how different coastal regions, coastal and marine ecosystems, and species will be impacted by climate change so that regional managers can plan appropriately. Although regional-scale climate projections are improving, their resolution remains low as a result of natural variability and local anomalies (Loder *et al.*, 2013). The question remains, how do we plan ahead when our vision of the future is uncertain.

In 2011, Fisheries and Oceans Canada (DFO) initiated the Aquatic Climate Change Adaptation Services Program (ACCASP) (DFO, 2014a). This program was developed in response to a globally recognized need to plan ahead using the information currently available even in the face of uncertainty (e.g., Helmuth *et al.*, 2014). One of the main goals of this program was to provide marine and fresh water managers with tools to help them adapt for climate change ("adaptation tools") (DFO, 2014a). ACCASP-led projects are being conducted in all bioregions. Planning ahead does not require that our view of the future is absolutely clear. Planning for the future should be precautionary, and managers should be aware of possible risks and changes that could appear. For this reason, the status quo scenario is often used: that global greenhouse gas emissions will not be significantly reduced in the near future.

1.2 CLIMATE CHANGE ADAPTATION TOOLS

As an initial scoping exercise under the ACCASP program, researchers conducted a risk assessment to identify the government divisions that may be at highest risk as a result of future climate changes in Canada (DFO, 2013b). Infrastructure damage was identified as a highly likely risk to DFO's Small Craft Harbours division. Next in line were species reorganization/displacement, changes in biological resources, and damage to ecosystems and fisheries. To understand localized risks to biological resources, and small craft harbours, assessment and modeling tools can be applied. Some climate change adaptation tools that are being developed in Canada, or have been developed elsewhere include Climate Change Vulnerability Assessments (CCVAs), and Species Distribution Models (SDMs).

Vulnerability Assessments (VAs) can be used to triage systems (i.e., harbours, coastline, coastal communities, or species) by their vulnerability to climate changes (i.e., their risk of damage, destruction, extinction, or loss of health and abundance). This can help identify the systems that will benefit from management efforts, and those that will not due to extreme vulnerability or relative robustness, so that managers can efficiently distribute often limited resources. CCVAs are continually evolving; there are many different approaches to the design and implementation of this tool (see for example, Williams *et al*., 2008; Chin *et al*., 2010; Mamauag *et al*., 2013; Young *et al*., 2010; Gaichas *et al*., 2014). In the case of species-level CCVAs, there has been a strong trend towards more quantitative, regional-scale approaches to improve their applicability to management adaptation programs and decision-making. This is because marine managers work at regional scales and because ranking of species by their relative vulnerability rather than broadly grouping them into high, medium, and low vulnerability categories may be more useful.

SDMs are quantitative, spatially explicit tools used to project species gain, loss, or shifting of fundamental habitat availability as a result of changing conditions. A combination of SDMs and CCVAs would be an invaluable addition to the climate adaptation toolbox. SDMs can provide more quantitative estimates of species exposure to potential climate change induced disruptions, while CCVAs can be used to assess the effects of species' life history, population, habitat and diet characteristics on overall vulnerability to climate change.

1.2.1 SDMs

Marine animals will react in various ways to climate-related changes. On a global scale, SDMs, sometimes called "bioclimate envelope models" (e.g., Cheung *et al*., 2009), have projected that species will shift towards higher latitudes and deeper depths; species invasions are projected to occur most toward the poles and local extinctions are projected to occur most in equatorial and polar regions (Cheung *et al*., 2009). These projections are made on the basis of global-scale climate projections, which some critique as being over-generalized. At smaller scales, the trajectory of climate change and therefore of species distribution shifts may not be poleward (e.g., Pinsky et al., 2013). Global projections, usually derived from a status-quo scenario, can be downscaled. Uncertainty in global projections is usually exacerbated at smaller scales due to regional variability. However, progress must be made in spite of uncertainty. Shackell *et al.* (2014) used downscaled warming projections for the northwestern Atlantic to project marine species shifts in habitat availability given observed temperature, depth, and location preferences. This provides useful information concerning how species' environmental preferences will guide their distributional responses to climate change. However, species interactions, adaptation, and dispersal will also impact marine species responses, both in distribution and population viability (Pearson and Dawson 2003; Guisan and Thuiller 2005). These traits can be assessed using a CCVA as a complement to an SDM.

1.2.2 History and evolution of Vulnerability Assessments

Vulnerability Assessments (VAs) have been used in various disciplines to assess the susceptibility of natural or human systems to negative impacts (vulnerability) as a result of human activities or natural pressures (Füssel and Klein, 2006). Often, the purpose of conducting a VA is to identify the most vulnerable systems so that measures can be taken to limit the stress imposed on those systems (e.g., Garthe and Hüppop, 2004; Füssel and Klein, 2006; Patrick *et al*., 2010; Stelzenmuller *at el*., 2010). The earliest VAs were designed in the early 1990s (e.g., Furst and Huffine, 1991). These VAs were largely descriptive, involved expert/mass opinion, and provided broad results. Ranking procedures for factors related to species/system vulnerability were developed to enhance VAs in the early 2000s (e.g., Stobutzki *et al*., 2001; Garthe and Hüppop, 2004). As our desire to lessen our impact on natural systems has increased, so too has our desire to improve VA models so they may more accurately assess all aspects of a species'/system's vulnerability. As a result, the framework of the ranking-style VA has shifted significantly over the past decade.

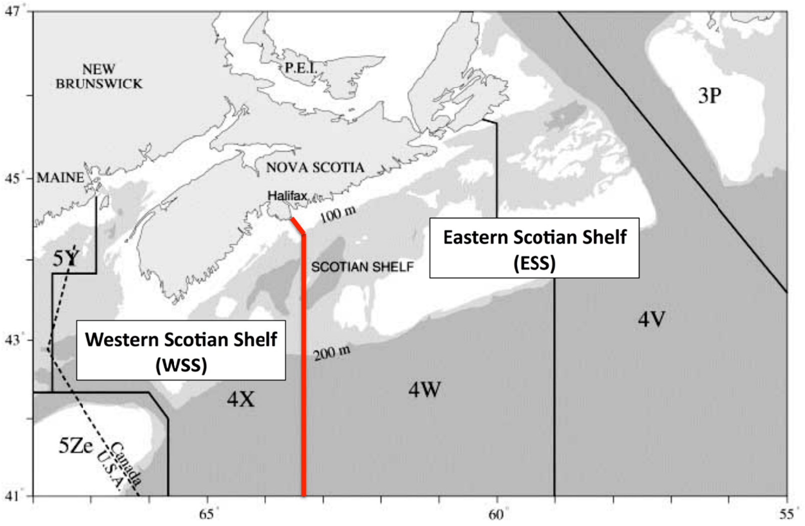
In 2001, Stobutski *et al.* (2001) developed a VA to assess fish species' vulnerability to overfishing. They suggested that a species' vulnerability is the product of its sensitivity and its adaptive capacity. By this it is meant that a species must be highly susceptible to negative impacts due to habitat, diet, etc., and also lack sufficient adaptive capabilities, in order to be considered highly vulnerable. This logic has been carried through to the most recently developed VAs. However, evaluation of the extent and magnitude to which species actually encountered the stressor in question was not included in the majority of earlier VAs, or was included as an additional spatial analysis. Exposure has been incorporated into more recent VAs (e.g., Chin *et al*., 2010; Mamauag *et al*., 2013; Gaichas *et al*., 2014). This change is in recognition of the fact that although a species may be sensitive and unable to adapt to a stressor, it cannot be vulnerable unless it is actually exposed to that stressor. This logic evolved as the tool became popular in climate change studies.

In 1990, the IPCC acknowledged VAs as potentially crucial tools for climate adaptation (IPCC, 1990). In the IPCC's first report (IPCC, 1990), they recognized the need to develop VA methodologies and other adaptation tools. In their 2001 report (IPCC, 2001), they acknowledged a need to enhance utility of Climate Change VAs

(CCVAs) for managers. In 2010, Chin *et al*. produced a CCVA that has set the base line for many CCVAs designed since. This CCVA evaluated vulnerability, of shark and ray species, as the product, through use of a conservative logic matrix, of exposure, sensitivity, and adaptive capacity. Within each of these categories, Chin *et al*. considered many species-specific traits, which have also been included in many CCVAs since. Young *et al*. (2010) took a similar approach, though more quantitative, to assess terrestrial species vulnerabilities to climate change on the scale of national parks and protected areas. However, these CCVAs have not included regional-scale, spatial, three-dimensional analyses of species exposure to climate change. In other words, the definition of exposure to climate change has been broadly defined according to global projections and generalizations concerning species habitats and interactions. In this thesis I refine the "exposure" factor by including an SDM and regionally downscaled projections of warming.

1.3 THESIS RATIONAL AND OBJECTIVES

Recognition of the limitations of SDMs and CCVAs when individually applied led to the idea that, together, these tools could be highly useful. SDMs can fulfill needs that CCVAs have yet to meet, and vice versa. A CCVA that includes a quantitative evaluation of species' exposure to climate changes may help marine managers make more informed decisions concerning resource prioritization for climate adaptation. The objective of this thesis was to design such a CCVA that can act as a template for other regions and be used to triage species of interest by their vulnerability to known climate drivers. In this thesis, use of the tool is piloted in the Scotian Shelf marine realm, off the coast of Nova Scotia, Canada (Figure 1.1), its utility is demonstrated at the population-level for a subset of species to make evident the importance of scale, and the question of how we govern for climate change under uncertainty is addressed. Only warming could be considered due to limitations in data and projections for other ocean climate drivers. Although this Vulnerability to Projected Warming Assessment (VPWA) only considers one aspect of climate change, though a very important aspect in the Scotian Shelf region, it represents one step further in the evolution of climate adaptation tools. Further, this is the first project of its kind under the ACCASP program. This work should guide continued CCVA evolution in other regions of Canada, and the globe. It is important to note that these tools can help managers adapt within a reasonable time frame, given likely scenarios of change. CCVAs are not meant to forecast exact time or magnitude of impact, but to identify areas of concern. Planning for the future and prioritizing resources given climate projections and CCVA results could prevent significant economic and social impairment from climate change in the future.



http://www.fao.org/docrep/003/x8985e/x8985e0a.jpg

Figure 1.1. Map of the Scotian Shelf taken from the Food and Agriculture Organization of the United Nations (http://www.fao.org/docrep/003/x8985e/x8985e0a.jpg), divided into western and eastern sections (WSS and ESS) according to the North Atlantic Fisheries Organization's (NAFO) fisheries divisions (i.e., 4X, 4W, 4V).

1.4 STRUCTURE OF THESIS DOCUMENT

Chapter 2 of this document encompasses a manuscript resubmitted in November 2014 following revisions for publication with the ICES Journal of Marine Science. This manuscript outlines the process of designing a VPWA, discusses its strengths and limitations, and demonstrates its utility through use of a case study, assessing the vulnerability of key marine species (33) and populations (WSS and ESS populations for seven out of 33 species) to projected warming on the Scotian Shelf, Canada (region shown in Figure 1.1). Results of the case study are also discussed with regards to their potential implications for regional marine management programs. Due to the nature of journal articles (i.e., that they require introduction, methods, results, discussion, and conclusion sections), there may be some overlap between Chapter 2 and other chapters, mainly Chapters 1 (Introduction) and 4 (Conclusion). Chapter 3 discusses the data needs and limitations of the VPWA developed in Chapter 2 as well as other CCVAs and SDMs, how the model could be improved to include climate stressors other than warming, how data limitations may prevent this from happening in the immediate future, and routes forward given currently available methods and data. Chapter 4 (the concluding chapter) discusses how the completed work fulfils the initial thesis objectives, the larger implications of the work, and how it contributes to growing regional, national, and global needs for climate adaptation tools.

CHAPTER 2 ASSESSING MARINE SPECIES VULNERABILITY TO PROJECTED WARMING ON THE SCOTIAN SHELF, CANADA

ABSTRACT

Resource managers need tools to prepare for biological changes resulting from climate change. We build on a popular tool, the Climate Change Vulnerability Assessment (CCVA), to identify vulnerable marine species. Only warming was considered, as warming is expected to have earlier impacts in the offshore than other climate drivers, and the field of acidification projections is not well developed. We present a generalized, semi-quantitative Vulnerability to Projected Warming Assessment (VPWA). We refine the typical "exposure" component to be a function of gain/loss of thermal habitat at multiple life stages. We also build on the traditional logic approach of CCVAs. We produce scores for each species, create a null distribution through Monte Carlo simulations, and use that distribution to designate vulnerability categories. We evaluate the vulnerability of 33 fish and invertebrate species, on the scale of the Scotian Shelf, Canada, to two warming scenarios, mild and severe, based on regional trends and projections. At smaller spatial scales, we evaluate populations of a subset of seven of these species. Populations in the southwest portion of the domain are found to be more vulnerable than those in the northeast. Overall, our results indicate that 53% of populations may be vulnerable under a severe (+3ºC) warming scenario, including currently endangered, threatened, and commercial populations (e.g. southwestern Atlantic cod, Smooth skate, Snow crab), while only 8% are vulnerable under the mild (+0.7 ºC) scenario (including Snow crab, and Sandlance). Populations triaged by relative vulnerability to regional warming should help managers prioritize resources and identify knowledge gaps. For this reason, and for its biological and ecological underpinnings, our method has broad relevance within the marine science and management field. As more information about other climate drivers and ecosystem-level consequences become available, our VPWA can be used as a stepping-stone in the continued development of CCVA methods.

2.1 INTRODUCTION

Warming, acidification, decreased oxygen, and increased freshwater input are some of the changes facing marine organisms as a function of anthropogenic climate change. The Intergovernmental Panel on Climate Change (IPCC) has made global-scale projections of how these changes may progress into the future under various emissions and assimilation rate scenarios (IPCC, 2013). Some uncertainty lies in the structure of these numerical models, the uncertainty of future CO_2 emissions, the lack of consideration or full understanding of natural variability in the global climate system, and the impossibility of hindcasting when current trends fall outside the range of past climate variations (Roe and Baker, 2007; Hawkins and Sutton, 2009; Planque *et al*., 2011; IPCC, 2013). At local scales, oceanographers are working to downscale these global projections. Due to a host of local-scale complications (e.g., ocean current patterns, topography, etc) and uncertainty in statistical downscaling models, uncertainty is exacerbated at regional scales (Chen *et al*., 2006; Brander *et al*., 2013; Bürger *et al*., 2013). As a result, most currently available regional climate projections are low resolution and highly uncertain (e.g., Loder *et al*., 2013). However, there is growing need for methods that will predict biological responses to climate change so that governments and community groups that are dependent on biological resources can plan climate adaptation strategies. There is also a growing understanding that such methods will have to be built and used regardless of current uncertainty regarding climate projections (e.g., Helmuth *et al*., 2014). Generalized, easy-to-use methods that use the best information available are needed. The assessment presented herein was, in fact, requested to help marine managers across Canada to translate scientific information for their uses in climate adaptation programs (see Appendix A for more details).

A commonly observed response of marine populations to climate change is a shift in distribution via migration (mobile species), larval dispersal (sessile species), or changes in abundance among regions (Parmesan and Yohe, 2003; Pearson and Dawson, 2003; Cheung *et al*., 2008; Pinsky *et al*., 2013). Species Distribution Models (SDMs), or bioclimate envelope models, have been used on both global (e.g., Cheung *et al*., 2009) and regional (e.g., Shackell *et al.,* 2014) scales to project shifts in marine species distributions or gain/loss of thermal habitat. In the case of Shackell *et al.* (2014), the most likely warming projections were used at a regional scale to project changes in distribution for 46 marine fish and invertebrate species. However, SDMs, in general, do not consider adaptive capabilities of individuals, populations, species, and ecosystems (Pearson and Dawson, 2003; Planque *et al.*, 2011). Traits reflecting adaptability and sensitivity can be evaluated using vulnerability assessments in addition to SDMs.

Vulnerability assessments are tools that describe the risk of harm to biological systems imposed by anthropogenic activities (Adger, 2006). Generally qualitative, vulnerability assessments evaluate various characteristics of populations, species, or communities that may increase or decrease vulnerability to a particular stressor. Recently, Climate Change Vulnerability Assessments (CCVAs) have been recognized as potentially useful tools for climate adaptation (Burton *et al.*, 2004; Fussel and Klein, 2006; Metzger and Schröter, 2006; IPCC, 2007; Williams *et al.*, 2008; Mamauag *et al.*, 2013; Small-Lorenz *et al.*, 2013). Many of the characteristics considered in CCVAs cumulatively describe the sensitivity and adaptive capacity of species/systems. In this way, CCVAs could act as important supplements to SDMs. While SDMs can provide estimates of future distributions and densities of marine populations, CCVAs can provide some insight into the future biological health (i.e., productivity and abundance) of populations as a result of projected environmental changes. It is important to identify both more vulnerable and less vulnerable species at regional scales so that management programs can adjust appropriately for future sustainability (White *et al.*, 2001; Metzger and Schröter, 2006; Gaichas *et al.*, 2014). It is especially important to identify those species that are most vulnerable to regional climate shifts so that other stresses to these species can be minimized or avoided (Brander, 2007).

Although methods vary among vulnerability assessments, the framework is usually quite similar. Generally, vulnerability is defined as some variation of the IPCC's definition: "[vulnerability is] the degree to which a system is susceptible to, or unable to cope with, adverse effects of climate change, including variability and extremes…[;] a function of the character, magnitude, and rate of climate variation to which a system is exposed, its sensitivity, and its adaptive capacity" (White *et al*., 2001, p. 21). Modern vulnerability assessments, for the most part, characterize population-level vulnerability to human-induced stressors by evaluating factors of exposure, sensitivity and, in some cases, adaptive capacity (Fussel and Klein, 2006).

Until recently (Gaichas *et al*., 2014), Climate Change Vulnerability Assessments (CCVAs) have been largely qualitative although it has been recognized by some that semi-quantitative CCVAs may be more easily applied to marine management programs, as they can potentially provide more refined vulnerability rankings (White *et al*., 2001; Williams *et al*., 2008). Gaichas *et al*. (2014) developed a semi-quantitative, community-level CCVA, but noted that species-level evaluations are also needed. In this paper, we develop a broadly-applicable, population-level vulnerability assessment (VA) that builds on the methods of Gaichas *et al*. (2014) and others (Füssel and Klein, 2006; Williams *et al*., 2008; Chin *et al*., 2010; Mamauag *et al*., 2013) to fill this niche. We take a novel approach by incorporating SDM projections into the assessment (using results from Shackell et al., 2014 for 33 species for which the additional information required for sull vulnerability assessment exist). We further improve the quantitative aspect by weighting model components by their importance to overall vulnerability as derived from literature. In this study, only warming was considered, thus we refer to our assessment methodology as a Vulnerability to Projected Warming Assessment (VPWA) from this point onward.

One of our main goals was to create a method that is generalized and explained in enough detail that managers and scientists from all regions could adapt it for their own climate adaptation purposes. Here we provide a detailed description of our VPWA design process and framework, discuss its applicability in other regions and our reasons for deviating from the qualitative line of CCVAs, and illustrate the utility of our VPWA via a case study. We also explore the tool's limitations, how auxiliary information can help interpret the results, and how the VPWA should be further developed as more information on projections and responses to additional climate drivers becomes available. We use the VPWA to rank the vulnerability (i.e., risk to future population health) of 33 commercially and/or ecologically important fish and invertebrate species to projected warming for the Scotian Shelf, Canada. We chose to evaluate spatially discrete populations (from the southern portion of the region, the western Scotian Shelf (WSS), and the northern portion of the region, the eastern Scotian Shelf (ESS)) of seven of these species to demonstrate the importance of scale to the outcome of the model.

Only warming is considered, as it is estimated to be the most influential, and most imminent driver in this region (Chabot *et al*., 2013), and reliable local-scale trends and projections do not yet exist for other climate change drivers (Loder *et al*., 2013). While it has been suggested that acidification may be an equally or more important driver of biological and ecological changes compared to warming (Orr *et al*., 2005; Turley and Gattuso, 2012), the field of forecasting pH spatial fields and associated biological responses is in its infancy. At present time, regional projections of pH change are low-resolution and two-dimensional (i.e., pH records are not available for depths below the surface). Further, our best assumption concerning the impacts of acidification on marine species is that shellfish and corals will be more greatly impacted than teleosts (e.g., Curran and Azetsu-Scott, 2012; Turley and Gattuso, 2012). For our purposes, the incorporation of acidification would not add any value to the VPWA framework due to the broad nature of this generalization. Further, the extent and magnitude of population-level exposures to acidification are currently immeasurable due to lack of high-resolution projections. Once more information becomes available, our framework could readily accommodate acidification effects.

This generalized, semi-quantitative, species/population-level VPWA, which incorporates SDM results and factor weighting, will act as a starting point for marine managers who are looking for directions on how to adapt for climate change. With the current state of climate change knowledge, contemplation over the effects of less certain impacts, such as acidification, and variables that may only impact certain species, such as increased susceptibility to disease (e.g., Lafferty *et al.*, 2004; Tlusty and Metzler, 2012), can follow use of the VPWA tool to triage species. Application of this VPWA should also help identify areas where more research is needed. As more data become available and projections are improved, a more complete range of climate drivers can be integrated into the framework, and more species can be assessed.

2.2 METHODS

2.2.1 Defining vulnerability

We follow the IPCC's definition of *vulnerability,* as provided earlier. Throughout the CCVA literature, definitions of vulnerability are very similar to the IPCC's definition. Definitions for the components of vulnerability (i.e., exposure [E], sensitivity [S], and adaptive capacity [AC]) have been discussed exhaustively in the field of CCVAs (Ionescu *et al*., 2008). The triad of E, S and AC is said to have its origin in human systems (e.g., Allison *et al*., 2009). In marine systems, there can be little distinction between sensitivity and adaptive capacity factors (Mark Nelson and Wendy Morrison, NOAA pers. comm.; Gardali *et al*., 2012). For this reason, we define sensitivity as a combination of what others have defined separately as sensitivity and adaptive capacity. Adapting appropriate existing definitions, we define exposure and sensitivity to projected regional warming as follows.

Exposure: The extent and magnitude to which a species' or population's surroundings will be negatively impacted by warming, i.e., the degree to which a species' or population's thermal habitat will be reduced due to warming. Species that will not lose any habitat due to warming or species that will gain habitat in response to warming are considered to have zero exposure.

Sensitivity: The degree to which a species or population may be negatively impacted, directly or indirectly, by exposure to warming. This includes the potential of a species or population to recover from adverse effects of warming, migrate toward more favourable conditions, and/or adapt to new conditions or ways of life. For our purposes, species or populations that are positively impacted by warming and/or highly adaptive were considered to have very low sensitivity.

2.2.2 Conceptual Framework

Our VPWA follows the vulnerability logic framework proposed by the IPCC (McCarthy *et al*., 2001). The logic is that a system can only be highly vulnerable to a stressor, in this case warming, if it is highly exposed, highly sensitive, and with insufficient adaptive capacity. Given our adapted framework, high vulnerability is the result of both high exposure and high sensitivity, which includes adaptive characteristics. This reflects a conservative approach that is prevalent in modern vulnerability assessment literature (Füssel and Klein, 2006; Chin *et al*., 2010; Magness *et al*., 2011; Mamauag *et al*., 2013; Reece *et al*., 2013; Wade *et al*., 2013; Gaichas *et al*., 2014). CCVAs are often structured within this framework, but how the components of vulnerability are evaluated varies. In the case of recent qualitative CCVAs, logic matrices have been used to derive vulnerability ranks from component (exposure, sensitivity, and adaptive capacity) ranks (e.g., Chin *et al*., 2010; Mamauag *et al*., 2013). In these cases, logic rules are also used to derive component ranks from species ranks for factors/traits/characteristics/attributes (hereafter referred to as "factors") that contribute to the components. These logic matrices embody the conservative approach discussed.

Prior to our study, we used a popular, recently developed CCVA matrix (Chin *et al*., 2010) to evaluate 33 marine populations on the Scotian Shelf (the same 33 species that we use in our case study summarized, herein). We found that the results were too broad and uninformative. As a result, we pursued a more quantitative approach, which would result in finer vulnerability ranks that could be more useful to marine managers in this region. For example, five species could be ranked as highly vulnerable, but it is important to then note their vulnerability relative to each other. To do this, we used a 1-5 scoring system and weighted factors (species or habitat-based traits related to exposure or sensitivity) within the assessment by their importance as derived from the literature. We also took a smaller-scale approach to evaluating exposure to warming by using an SDM to project adult thermal habitat availability in the region, and predicting loss of regional thermal habitat at larval and spawning life stage. However, the conservative approach taken by Chin *et al*. (2010) is reflected in our semi-quantitative model where vulnerability is the product of exposure and sensitivity. Multiplying the two components ensures that high vulnerability scores are only obtained if scores for both exposure and sensitivity are high.

2.2.3 The VPWA

Several common themes in the climate change vulnerability literature were considered when building our VPWA. One theme is that loss of habitat due to warming is a major determinant of marine species vulnerability (Davies *et al*., 2004; Poloczanska *et al*., 2009; Gilly *et al*., 2013; Hollowed *et al*., 2013). A second is that early life stages are often the most vulnerable to environmental change (Cushing, 1997; Pörtner and Farrell, 2008; Mantzouni and MacKenzie, 2010; Petitgas *et al*., 2013). A third theme is that healthy, often r-selected and mobile populations are more resilient to the stress caused by environmental change because of fast inter-generational adaptation and mobility (Musick, 1999; Davies *et al*., 2004; Brander, 2007; Mora *et al*., 2007; Hsieh *et al*., 2008; Planque *et al*., 2010). A fourth theme is that vulnerability cannot be characterized by only one or two biological or habitat-based traits (Davies *et al*., 2004). These themes guided the population of our VPWA framework with factors, which are defined as population-level traits or circumstances that relate to its exposure or sensitivity to warming. Species were given a score for each. Factors related to these common themes are also present in other recently developed CCVAs (Chin *et al*., 2010; Pecl *et al.,* 2011; Reece *et al*., 2013; Gaichas *et al*., 2014).

It is widely acknowledged in vulnerability assessment literature that some factors influence vulnerability more strongly than others (Downing and Patwardhan, 2005; Hiddink *et al*., 2007). For this reason, factors were weighted by their importance as reported in the literature, as well as by data availability, so as to limit the impact of uncertainty on the outcomes of the model. We judged a factor as "data-limited" if no data could be found to obtain scores for said factor for more than 25% of our study species. Factors were given a level of importance between 1 and 3, where 3 was applied to the most important factors with strong data, 2 was applied to factors of medium importance or highly important data-limited factors, and 1 was applied to the least important factors or data-limited factors of medium importance. We recognize that the numbers chosen to weight factors are arbitrary, but they should be considered relative. Our 1-3 method allowed us to weight our factors relative to each other to ensure the effect of each factor on the outcome of the model reflected this relative importance. Further, a no differential weighting among factors can prove misleading, The impact of highly important factors may be dampened by other factors that are less important but weighted the same. The exposure (E1 – E3) and sensitivity (S1 – S7) factors included in this VPWA are described, and their importance weightings justified, below.

Factors of Exposure:

E1 = Projected adult habitat loss (Shackell et al., 2014); Importance=3.

E2 = Projected loss of larval temperature window; Importance=2.

E3 = Projected loss of spawning temperature window; Importance=1.

In many other CCVAs, a population's exposure is evaluated as high, medium, or low depending on its habitat type (i.e., pelagic, coastal, demersal, etc.) and the extent to which those habitats are predicted to experience climate change. We took this a step further by quantifying exposure as the predicted amount of Scotian Shelf habitat lost given regional warming projections. Species/populations which are projected to gain local thermal habitat or for which thermal habitat will remain the same, were given the lowest exposure score, 1 (Figure 2.1). This analysis was done, for the adult phase, through use of an SDM (Shackell *et al*., 2014) that considers species temperature and depth preferences as well as current location, which is often tied to habitat type and species interactions (Pearson and Dawson, 2003). SDM results for the 14 discrete populations (two populations per species for a subset of seven species) are included in Appendix B.

Factor E1 was weighted as highly important because adults often determine species distribution (i.e., the "realized niche") (Pearson and Dawson, 2003; Wake *et al*., 2009), and population health (e.g., Davies *et al*., 2004). Additionally, confidence in SDM results is highly dependent on data availability and catchability of the species (Shackell *et* *al*., 2014). Due to limited spatial data for larval and spawning stages of marine species, we could not use an SDM to project habitat loss for these life stages. As a result, these factors were weighted less heavily. As more data were available for larval stages than for spawning stages regarding temperature preferences, E2 (27% of species completely lacking data) was weighted more heavily than E3 (58% of species completely lacking data). These factors were included because early life stages (spawning, egg, and larval stages) often act as a "bottleneck" to species vulnerability to environmental change (Cushing, 1997; Pörtner and Farrell, 2008; Mantzouni and MacKenzie, 2010; Petitgas *et al*., 2013); these life stages often tolerate a narrower range of temperatures. In the case of egg and larval stages, this is a function of the fact that these stages often have a higher surface area to volume ratio, have less ability to choose their surroundings, and are more subject to other stressors (e.g., predation). In the case of the spawning life stage, this is a function of the fact that spawners choose the optimal conditions for their eggs.

For factors E2 and E3, each species' larval or spawning temperature window was obtained from literature. The proportion of that window that would not occur in the region of interest if warming progressed as expected was calculated and translated into a factor score. This method is visualized in Appendix C, and references are provided in Appendices E and F.

Factors E1 and E2 were given exponential weightings. This was due to the suggestion that loss of habitat is the initial driver of species vulnerability to climate change, where shifting environmental conditions can force species to move, adapt, or perish (e.g., Perry *et al*., 2005). With this assessment, vulnerability is determined by how sensitive a species may be to a certain level of exposure. Habitat loss at all life stages is important, but in order to reduce the impact of uncertainty on the results of our model, only factors with greater data availability, E1 and E2, were given exponential weightings. Factor E3 was given a weighting of 1 due to severe data limitation and the consideration that larval stages are often more sensitive to environmental change. Given the "bottleneck" effect that the larval stage can have on species vulnerability (e.g., Petitgas *et al*., 2013), factor E2 would have been given a weighting of 3, if not for significant (>25%) data limitation.

Factors of Sensitivity:

S1 = Latitudinal range; Importance=2.

S2 = Habitat specificity; Importance=2.

Factors of sensitivity determine how negatively impacted a species could be as a result of regional habitat loss (exposure). If the species is found over a large latitudinal range, it is less likely to suffer from loss of habitat on the Scotian Shelf alone. This is a highly important factor, but it is important to note whether a species is found over a wide range of habitat types throughout its latitudinal range, or found only in specific, rare habitats. Habitat specificity will determine whether a species will be able to find suitable habitat when forced out of current locations, but latitudinal range will determine the rate at which the entire species/population will be impacted by habitat loss (i.e., exposure). As these two factors are equally important, they were weighted the same. However, a long branch of literature supports the importance of life history and population status (i.e., exploitation history) to the sensitivity of populations to additional stressors (Hutchings, 2002; Davies *et al*., 2004; Brander, 2007; Planque *et al*., 2010), whereas habitat does not appear as often as a main driver of population health. Therefore these factors were given a weight of 2.

S3 = Diet Specificity; Importance=1.

Because marine species of the North Atlantic Ocean are often characterized as "generalist feeders" (Smith and Link, 2010), diet is often not as important to species/population sensitivity as habitat specificity, range, life history, or exploitation history. For this reason, this factor was given a weighting of 1. We acknowledge that the region of study influenced this choice; weighting of this and other factors should be re-visited when using the VPWA in different regions, where species with diet specificity exist.

S4 = Population status; Importance=3.

It is generally accepted that exploitation history and life history are the largest determinants of a population's health, and ability to cope with negative impacts and adapt to new conditions (Davies *et al.*, 2004; Hutchings and Reynolds, 2004; Brander, 2007; Mora *et al.*, 2007; Planque *et al.*, 2010; Hsieh *et al.*, 2008; Shackell *et al.*, 2012; Niiranen *et al.*, 2013). Therefore, population status, as reflective of exploitation history, was inferred to be the most important determinant of species sensitivity to warming. Overexploited species generally have reduced population health, meaning reductions in reproductive potential, distribution, and size at maturity (Hutching and Reynolds, 2004; Brander, 2007). This weakens a population's ability to cope with changes individually and through generational evolution (Planque *et al.*, 2010). For this reason, factor S4 was given a weight of 3.

For factor S4, we translated the Committee on the Status of Wildlife in Canada's (COSEWIC) and Wild Species's categories of risk into scores. COSEWIC is a government body dedicated to the assessment of population health for both marine and terrestrial species in Canada (COSEWIC, 2012), and Wild Species is a branch of COSEWIC that provides "working statuses" for species not assessed by COSEWIC. COSEWIC uses local data to evaluate changes in biomass, number of individuals, and geographic range over time. The degree of negative change determines whether a population is categorized by COSEWIC as "Not at Risk", "Special Concern", "Threatened", "Endangered", "Extirpated" (not considered here because irrelevant to location-specific study), or "Extinct" (not considered here) (COSEWIC, 2012). Where COSEWIC attempts to categorize a population but data are greatly limited, it is considered "Data Deficient" or given an estimated status through *Wild Species* (Wild Species, 2012). *Wild Species* categories are "Secure" (here, given a score of 1 for this factor, equivalent to Not at Risk species), "Sensitive" (here, considered to be less sensitive than Not at Risk species but more sensitive than special concern species; given a score of 2), "May be at Risk" (here, given a score of 3, equivalent to special concern species), and "Not Assessed". We considered those species that were not assessed by either organization to have a rank equivalent to "Data Deficient" and "Sensitive" species. This was to ensure that we did not severely underestimate a species' sensitivity due to a lack of interest, concern, or data. Often, species are not assessed due to evaluated high abundance through fishery stock assessments. However, a history of overexploitation on the Scotian Shelf (e.g., Shelton *et al*., 2006) lead us to score "Not Assessed" commercially exploited species conservatively for this factor.

We chose an assessment program specific to Canada to ensure that population-level concerns were captured. The International Union for Conservation of Nature (IUCN) red list evaluates species population status in a similar way to COSEWIC and should be internationally relevant (IUCN, 2014). IUCN red list categories also fit the 1-5 scale, so this would be a suitable alternative for managers in other regions where there is no local assessment program.

S5 = Inherent resilience; Importance=2.

Resilience to change is determined by a species' generation time, size and age at maturity, and fecundity (Musick, 1999). This is another highly important determinant of adaptability and health, and therefore sensitivity (Winemiller and Rose, 1992; Hutchings, 2002). Species with high fecundity and generation time can adapt to climate change more easily through generational evolution. Long-lived species that reproduce infrequently cannot adapt quickly. These are broadly applicable biological concepts. Population status can have a large impact on resilience through its reduction in reproductive capacity and size at maturity (e.g., Planque *et al*., 2010). Recordings of species' generation time, size and age at maturity, and fecundity may be outdated if populations have been heavily exploited since the recordings were made. For this reason, resilience was weighted less heavily than population status. Resilience rankings were found on Fishbase.org (Froese and Pauly, 2014), or were estimated using the instructions given in Musick (1999) based on measures of fecundity, generation time, size at maturity, etc. It was assumed that the biological concept could be applied to invertebrates as well as to fish. For example, literature states that Northern Pink Shrimp can live to a maximum of 5 years in the Gulf of Maine, which is close to the Scotian Shelf. According to Musick (1999), a maximum age between 4 and 5 years corresponds to medium resilience. Musick's (1999) resilience ranks of very low, low, medium, and high, correspond to sensitivity scores of 5, 3.6, 2.3, and 1 respectively (Figure 2.1); therefore Northern Pink Shrimp received a score of 2.3 for this factor (Appendix E).

S6 = Adult mobility; Importance=2.

S7 = Larval dispersal capacity; Importance=1.

Adult mobility is a large determinant of whether a species or population can escape unfavourable conditions and search for more appropriate habitats. This factor is important, but not as important as population status; therefore it was given a weighting of 2. Where adults often determine population distribution, there are cases where larval dispersal plays a crucial role in the colonization of new habitats. In the case of many sessile animals, this is the only way for populations to change distribution. Factor S7 takes this into consideration. However, larvae are often at the mercy of wave action, currents, and extreme weather; therefore, dispersal may not be as effective as adult migration; adults have more control over where they migrate. For this reason, this factor was given a weight of 1.

2.2.4 Scale

A scale of 1-5 was chosen for the factor scores (Figure 2.1); each species received a score between 1 and 5 for each factor. This decision was inspired by the Likert scale (Likert, 1932), and other vulnerability assessments (e.g., Stelzenmüller *et al*., 2010). Although the choice of scale is subjective, we found that a scale of 1-5 was easy to comprehend, allowed for a mid-range score (i.e., 3) to apply where uncertainty is high (e.g., Garland, 1991), and was easy to apply to all factors included in the VPWA. For the purpose of consistency, the same scale was used for all factors. Qualitative factors required integer scores between 1 and 5, and quantitative factors required a continuous scale between 1 and 5 to a decimal place of 2.

2.2.5 Model structure

For each component of vulnerability (exposure and sensitivity), factor scores, weighted by importance (factor score multiplied by or given the exponent of the importance rank), were summed for each species (Figure 2.1). Recall that factors E1 and E2 were given exponential weightings due to their acting as main drivers of species vulnerability. The assumption is that these factors are independent of one another, and that their impact on the components of vulnerability is additive, not synergistic. Although some factors may be interrelated in some respects, each factor is uniquely important. Final component scores were multiplied to obtain a vulnerability score for each species. As previously discussed, the components were multiplied rather than summed because vulnerability depends on the interaction of the components (i.e., they are not independent (Davies *et al*., 2004)), and to reflect a conservative approach. Each species vulnerability score was then scaled between 0 and 1 for ease of comprehension. All computations were done using R Statistical language (R Core Team, 2012).

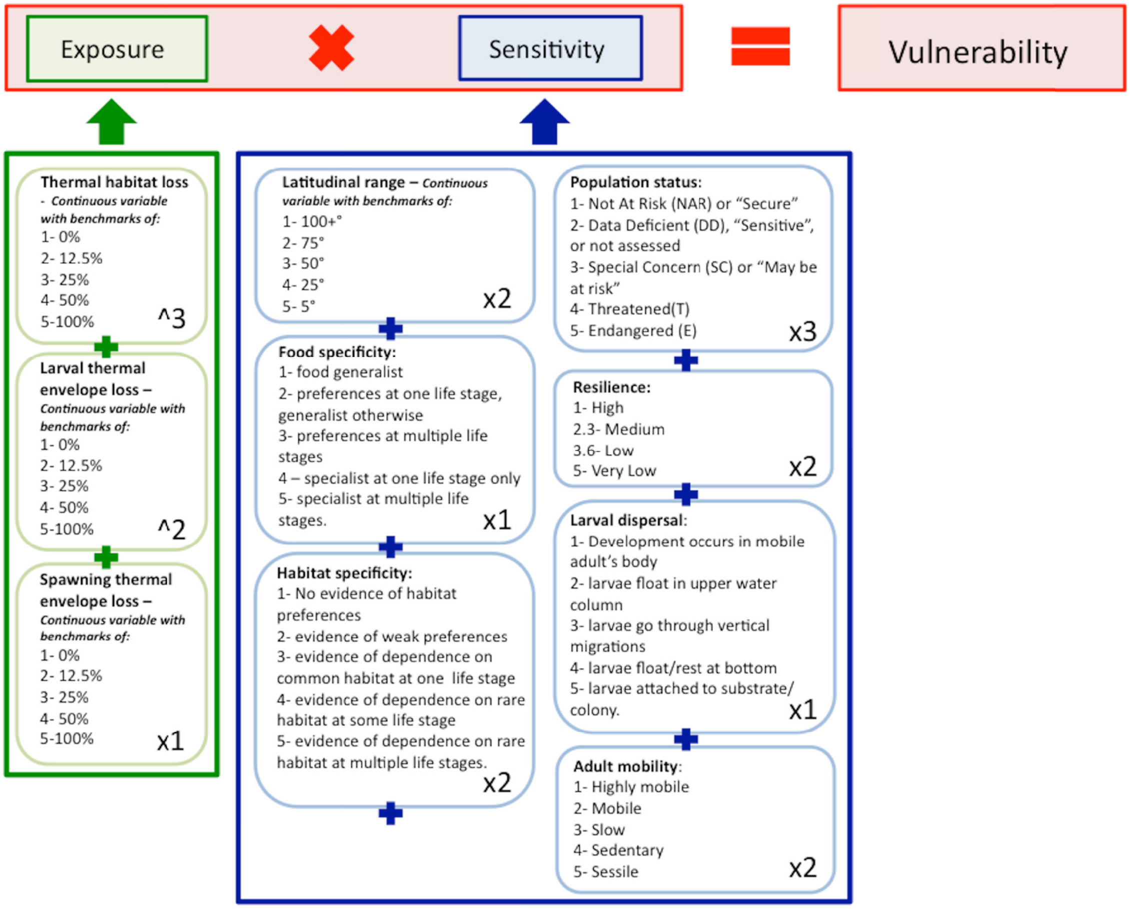


Figure 2.1. VPWA framework developed and used in this study. This VPWA follows the general framework of other modern vulnerability assessments and evaluates factors that reflect common themes in climate change literature. In green are factors of exposure, and in blue are factors of sensitivity. Factors are weighted and scores are combined as shown; a species obtains a score between 1 and 5 for a factor, this score is multiplied by or given the exponent of the importance weighting of that factor, all weighted factor scores are summed to obtain a component (exposure or sensitivity) score, and then component scores are multiplied to obtain a vulnerability score. Vulnerability scores are scaled from 0-1 for ease of comprehension.

2.2.6 Testing the VWPA

It is common practice to test vulnerability assessment models using a sensitivity analysis (Oakley and O'Hagan, 2004). A sensitivity analysis evaluates the sensitivity of the model response to each individual parameter, that is, the effect that each factor has on the final outcome of the model (i.e., the vulnerability score). The purpose of this analysis is to identify factors that are having an effect on the outcome of the model that is disproportionate to their level of importance, and factors that are irrelevant or redundant, such that they have little effect on the outcome of the model. A sensitivity analysis was conducted on the VPWA using the package "pse" R statistical language (Chalom and Lopez de Prado, 2012; 2013; Appendix D). The function "LHS" (Latin Hypercube Sampling for uncertainty and sensitivity analyses) was used to create a hypercube of our model. Factor scores between 1 and 5 were generated randomly. The function "plotprcc" was used to measure the partial correlation coefficient of each factor. Prcc values and p-values indicated that all factors have a significant impact on the model response, the degree of which reflects their weighting (Table D.1 and Figure D.1).

2.2.7 Ranking species by vulnerability score

Due to the conservative nature of the model, the distribution of all scores possible is positively skewed (Figure 2.2). This distribution was obtained via a Monte Carlo simulation (Mooney, 1997) of our VPWA. Tertiles of this distribution can be used to divide scores into low, medium, and high vulnerability categories. Taking a precautionary approach (RioDEC, 1992; De Sadeleer, 2007), species in both medium and high categories were considered to have notable vulnerability.

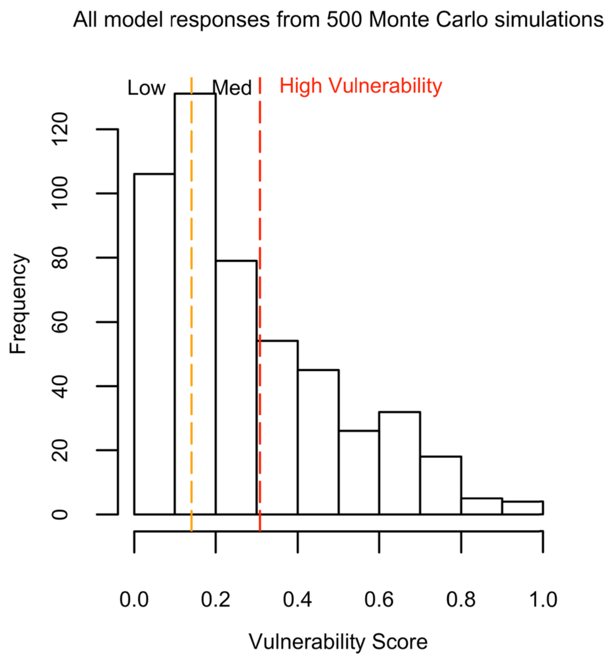


Figure 2.2. Histogram of all vulnerability scores possible with our VPWA. These scores were generated via a Monte Carlo simulation of our VPWA model (n=500) where scores between 1 and 5 (1, 2, 3, 4, or 5 in the case of discrete factors) were generated randomly for each factor.

2.2.8 Managing Uncertainty

Some species are better studied than others. To reflect differences in data availability and concurrence among species, we adopted the IPCC's guidelines for evaluating certainty (Risbey and Kandlikar, 2007; Figure 2.3). We evaluated our confidence in factor scores based on the quantity of relevant literature and the agreement among authors. Where species were greatly lacking in data or agreement (one reference or no references for more than two factors, or greater than 40/60 split in literature opinion), these species were not assessed. For the Scotian Shelf case study, these species included Barndoor Skate (*Dipturus laevis*), Jonah Crab (*Cancer borealis*), Rock Crab (*Cancer irroratus*), Sea Raven (*Hemitripterus americanus*), and Hagfish (*Myxine glutinosa*). Where a species had no references or significant disagreement for one or two factors, it was given a neutral score (i.e., 3) for that/those factor(s) in order to not over- or under-estimate vulnerability (Garland, 1991; Appendix E).

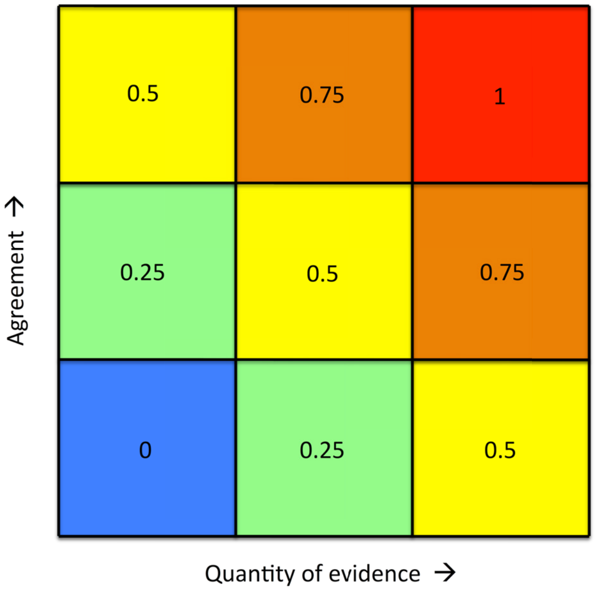


Figure 2.3. Method to evaluate certainty (0-1, 0 representing 0% certainty and 1 representing 100% certainty) when conclusions are drawn from literature (adapted from Risbey and Kandlikar, 2007). Certainty is a function of the quantity of literature supporting the conclusions and the agreement among literature sources. These ratings were used to determine importance weightings of factors (if a factor was highly uncertain for more than 25% of case study species, it was given a lower weighting) and to determine whether species could be accurately assessed with our VPWA or not (species with low certainty for more than 2 factors were not assessed).

2.2.9 The case study

To illustrate the utility of this VPWA, we used a case study of marine species found on the Scotian Shelf, Canada. Only warming could be considered as other climate projections are less developed in this region. Thirty-three local species were assessed using the VPWA. Some of these are managed separately in the wester and eastern portions of the Scotian Shelf according to fisheries divisions (designated by the North Atlantic Fisheries Organization, NAFO; Figure 1.1); these are refered to as WSS and ESS populations, and were evaluated separately in this analysis (Table 2.1). Factor scores for species/populations were derived from a robust literature review (all factors), spatial data (e.g., S1), and/or predictive modeling (e.g., E1, E2, and E3). For all species/populations' factor scores and corresponding references, please see Appendices E and F.

Table 2.1. Thirty-three species, including WSS and ESS populations of a subset of seven of these species, assessed using the VPWA. These species were grouped by functional group (as in Shackell and Frank, 2007), commercial status (C=Commercial, and N=Non-commercial), and population status (S= "sensitive", N.A.=not assessed, E=Endangered, N.R.=Not at Risk or "secure", S.C.=Special Concern or "may be at risk", and T=Threatened). Categories matching acronyms were obtained from the Committee on the Status of Endangered Wildlife in Canada (COSEWIC, 2012), and categories in quotations were obtained from Wild Species (Wild Species, 2012) when the species was not assessed by COSEWIC.

Common name	**Scientific name**	**Functional Group**	**Commercial Status**	**Population Status**
American Lobster	*Homarus americanus*	Decapod	C	N.A.
American Plaice	*Hippoglossoides platessoides*	Large benthivore	C	E
ESS Atlantic Cod	*Gadus morhua*	Piscivore	N	E
WSS Atlantic Cod	*Gadus morhua*	Piscivore	C	E
Atlantic Herring	*Clupea harengus*	Planktivore	C	N.R.
Atlantic Wolffish	*Anarhichas lupus*	Large benthivore	N	S.C.
Blackbelly Rosefish	*Helicolenus dactylopterus*	Medium benthivore	N	N.R.
Capelin	*Mallotus villosus*	Planktivore	N	N.R.
ESS Cusk	*Brosme brosme*	Piscivore	N	E
WSS Cusk	*Brosme brosme*	Piscivore	N	E
ESS Haddock	*Melanogrammus aeglefinus*	Large benthivore	C	N.R.
WSS Haddock	*Melanogrammus aeglefinus*	Large benthivore	C	N.R.
Halibut	*Hippoglossus hippoglossus*	Piscivore	C	N.R.
Little Skate	*Leucoraja erinacea*	Medium benthivore	N	S
Longhorn Sculpin	*Myoxocephalus octodecemspinosus*	Medium benthivore	N	N.R.
Monkfish	*Lophius americanus*	Piscivore	C	S
Moustache Sculpin	*Triglops murrayi*	Medium benthivore	N	N.R.
Northern Shortfin Squid	*Illex illecebrosus*	Piscivore	C	N.A.
Northern Shrimp	*Pandalus borealis*	Planktivore	C	N.A.
Ocean Pout	*Zoarces americanus*	Medium benthivore	N	S
ESS Offshore Hake	*Merluccius albidus*	Piscivore	N	N.A.
WSS Offshore Hake	*Merluccius albidus*	Piscivore	N	N.A.

ESS Pollock	*Pollachius virens*	Piscivore	C	S
WSS Pollock	*Pollachius virens*	Piscivore	C	S
ESS Red Hake	*Urophycis chuss*	Piscivore	C	N.A.
WSS Red Hake	*Urophycis chuss*	Piscivore	C	N.A.
Redfish spp.	*Sebastes species*	Piscivore	C	E
Sandlance spp.	*Ammodytes spp.*	Planktivore	N	N.R.
ESS Silver Hake	*Merluccius billinearis*	Piscivore	C	N.R.
WSS Silver Hake	*Merluccius billinearis*	Piscivore	C	N.R.
Smooth Skate	*Malacoraja senta*	Medium benthivore	N	S.C.
Snow Crab	*Chionoecetes opilio*	Decapod	C	N.A.
Spiny Dogfish	*Squalus acanthias*	Piscivore	C	T
Thorny Skate	*Amblyraja radiata*	Large benthivore	N	S.C.
Turbot	*Reinhardtius hippoglossoides*	Piscivore	C	N.R.
White Hake	*Urophycis tenuis*	Piscivore	C	T
Windowpane	*Scophthalmus aquosus*	Medium benthivore	N	N.R.
Winter Flounder	*Pseudopleuronectes americanus*	Medium benthivore	N	N.R.
Winter Skate	*Leucoraja ocellata*	Large benthivore	N	T
Witch Flounder	*Glyptocephalus cynoglossus*	Medium benthivore	C	S

Each species or population was assessed twice, once given warming and SDM projections for the year 2030, and once for the year 2060 (all species have projected habitat availability from Shackell *et al*. (2014); projections for WSS and ESS populations are given in Appendix B). Scores for exposure factors were different between projections, but all other factor scores, due to their biological nature, remained the same (Appendix E). Warming of 0.7ºC was projected for waters to 100m, and 0.35ºC for waters below 100m for the year 2030, given SST empirical trends since 1985. Warming of 3ºC for waters to 100m and 1.5ºC for waters below 100m was projected for the year 2060 given the mid-range of ensemble AR5 ESM August SST projections for the area (Shackell *et al*., 2014). These were the best projections available for our region at the time our research was conducted (see Shackell *et al*., 2014 for more information on uncertainty related to these projections). Where the projections of habitat availability from Shackell *et al*.'s SDM were highly uncertain, this uncertainty was noted as high for the appropriate species for the habitat availability factor (E1).

Although the warming projections used were for the years 2030 and 2060, it is important to note that the warming expected for these years could occur much earlier as a result of natural variability and related uncertainty in model projections (Loder *et al*., 2013). For example, the temperatures expected for the year 2060 were experienced on the Scotian Shelf in the year 2012 and documented as "record highs" (Galbraith *et al*., 2013). For this reason, we do not refer to these warming scenarios by the projection years, we refer to them, from here on, as the mild (+0.7ºC SST) and severe (+3ºC SST) warming scenarios. Of course, these projections are mild and severe relative to each other, but this language was used to prevent the illusion that our results provide a 2030 or 2060 "time stamp" for population vulnerability. It is important to acknowledge this as managers generally plan on shorter time frames (e.g., Füssel, 2007).

The species assessed were chosen because they are one or more of the following: ecologically important (i.e., highly common prey/predator species and species that are dominant within their respective functional groups (e.g., Gaston and Fuller, 2008)), commercially important, and/or a current conservation priority, i.e., currently given a population status of Special Concern, Threatened, or Endangered by COSEWIC (COSEWIC, 2012). This was to ensure that the results would be relevant to a wide variety of ocean management divisions. Data availability was also a determinant. Species and populations were grouped by functional group (refer to Shackell and Frank, 2007 for details on size and habitat-defined functional groups), commercial status, and population status so that vulnerability scores could be compared among groups (Table 2.1). The Kruskal-Wallis test was used to compare vulnerability scores among functional groups, commercial status groups, and population status groups.

2.3 RESULTS

2.3.1 The Case Study: Scotian Shelf, Canada

Under the mild warming scenario, only 2.5% of species have vulnerability scores in the upper two tertiles: only Moustache Sculpin (*Triglops murrayi*) (Figure 2.4). Under the severe warming scenario, 45% of species have notable vulnerability. Under both scenarios, western Scotian Shelf (WSS) populations are more vulnerable than eastern Scotian Shelf (ESS) populations, except in the case of Pollock (*Pollachius virens*). The maximum vulnerability score for the severe scenario is more than double that of the mild scenario (Figure 2.4) due to greater projected loss of thermal habitat.

Commercial perspective

The Kruskal-Wallis test indicated that, under the severe warming scenario, the vulnerability scores of currently commercially-fished species are significantly (p=0.005) lower than those of non-commercial species on average (Figure 2.5). The difference given the mild scenario is also significant (p=0.007). However, under the mild warming scenario, no commercial species have vulnerability scores above the low vulnerability threshold (Figure 2.4). Under the severe scenario, Snow Crab, ESS Pollock, WSS Pollock, WSS Cod, and WSS Red Hake have high vulnerability scores (Figure 2.4).

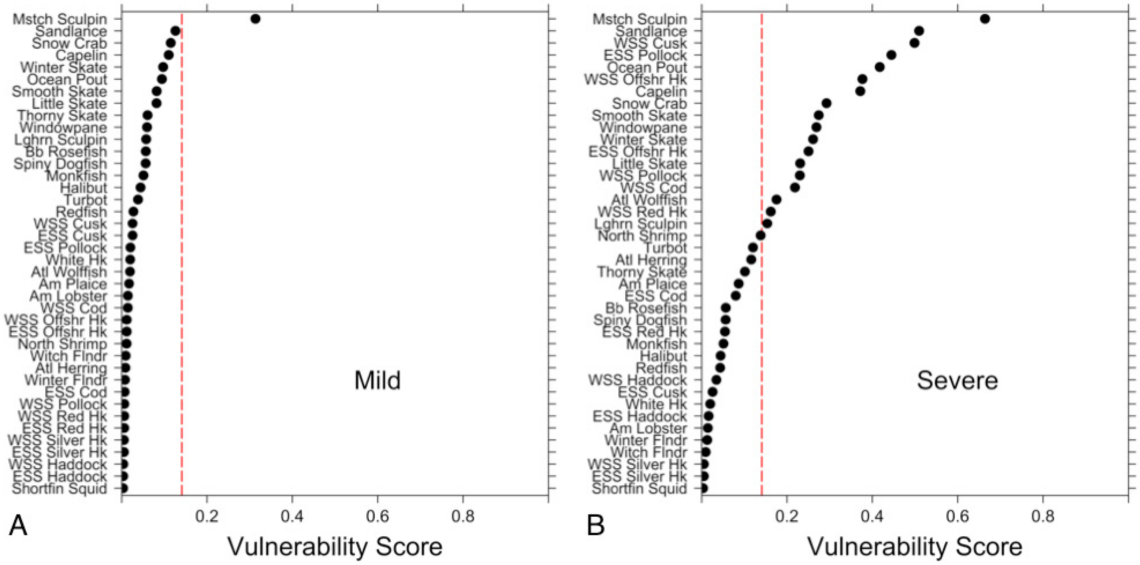


Figure 2.4. Vulnerability Scores of study species under both mild (A) and severe (B) warming scenarios. The red dotted line depicts the boundary between the first (low vulnerability) and second (medium vulnerability) tertiles of the model's random distribution. As discussed in the text, species in both high and medium vulnerability categories (second and third tertiles) are considered vulnerable (all species to the right of the red dotted line). All species falling to the left of the line are not considered to have notable vulnerability.

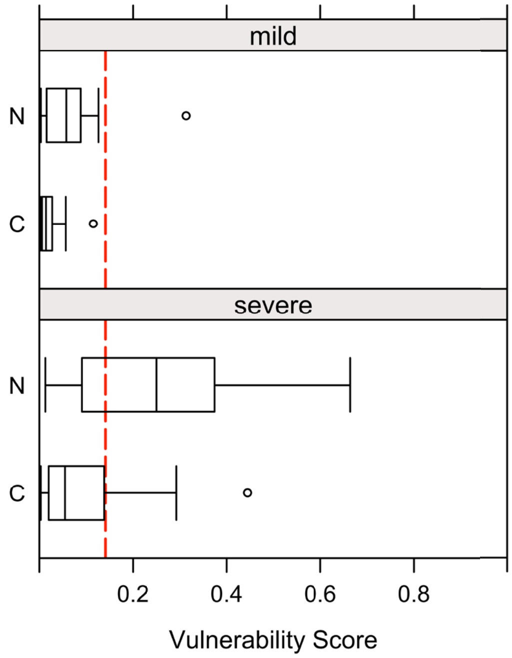


Figure 2.5. Average vulnerability scores of commercial (C) and non-commercial (N) species under the mild (top) and severe (bottom) scenarios. The red line represents the division between the first (low) and second (medium-high) vulnerability tertiles.

Species at risk perspective

A Kruskal-Wallis test indicated that vulnerability scores do not differ significantly among species of differing population statuses for either warming scenario (p=0.901 and p=0.356 for severe and mild scenarios respectively). However, it is noteworthy that more Not Assessed, and Not at Risk species/populations are vulnerable than Endangered, and Threatened species/populations (Figure 2.6). Under the mild warming scenario, the most highly vulnerable species is Not at Risk (Moustache Sculpin). Under the severe warming scenario, two of three Special Concern, one of three Threatened, three of six Sensitive, three of eight Not Assessed, two of five Endangered, and five of fourteen Not at Risk species have high vulnerability scores (Figure 2.4).

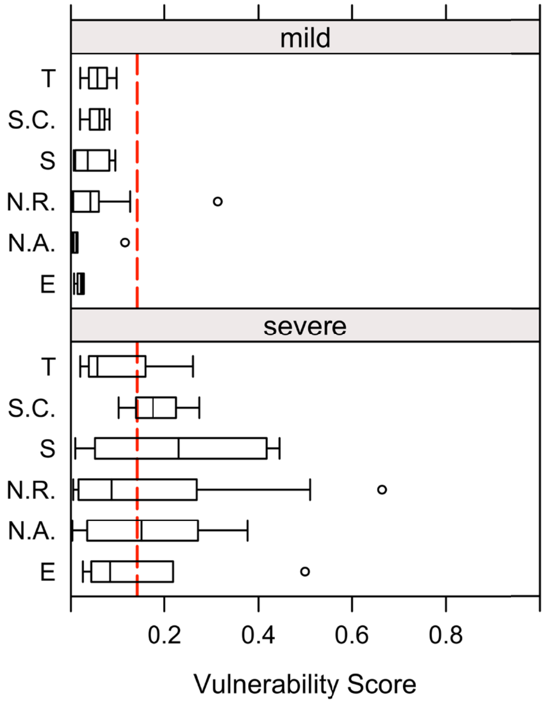


Figure 2.6. Average vulnerability scores of study species grouped by COSEWIC population status under the mild (top) and severe (bottom) scenarios. The red line represents the division between the first (low) and second (medium-high) vulnerability tertiles. (T: Threatened (COSEWIC, 2012), S.C.: Special Concern (COSEWIC, 2012) or 'may be at risk' (Wild Species, 2012), N.R.: Not at Risk (COSEWIC, 2012), or 'secure' (Wild Species, 2012), E: Endangered (COSEWIC, 2012), S: 'Sensitive' (Wild Species, 2012), N.A.: Not Assessed).

Ecosystem perspective

The Kruskal-Wallis test indicated that, on average, vulnerability scores do not differ significantly (p=0.424) among functional groups under the severe scenario, but did differ under the mild scenario (p=0.076) (Table 2.1). Under the mild warming scenario, vulnerability scores between piscivores and medium benthivores differ significantly (p=0.021) on average under the mild scenario. Most piscivores have very low vulnerability scores, while medium benthivores have higher scores on average, with one species, Moustache Sculpin, having the highest relative score under both warming scenarios (Figure 2.4). Figure 2.7 shows that the functional group with the most vulnerable species is the planktivores, and the functional group with the least vulnerable species is the piscivores. Two of four planktivores have scores in the upper two tertiles under the severe scenario (Sandlance and Capelin). Six of nine medium benthivores are vulnerable under the severe scenario. Of these six, only Moustache Sculpin is has a high vulnerability score under the mild scenario (Figure 2.4). In contrast, all other functional groups have at least 50% of species falling within the low vulnerability category under the severe scenario.

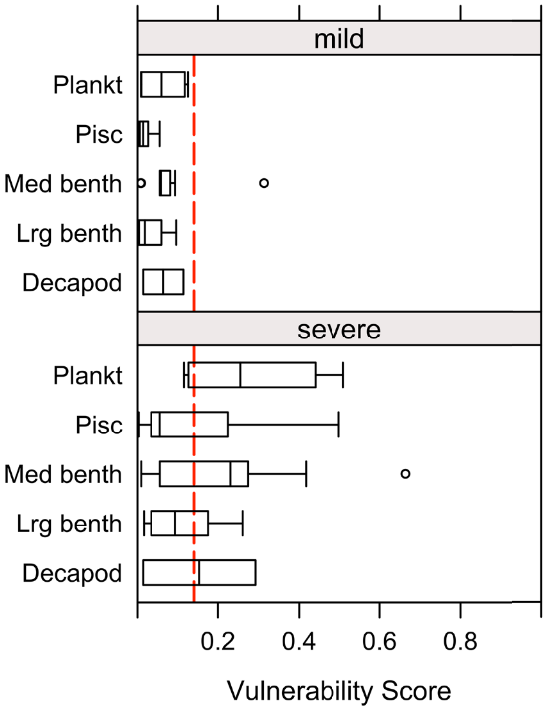


Figure 2.7. Average vulnerability scores of species grouped by functional group under the mild (top) and severe (bottom) scenarios. The red line represents the division between the first (low) and second (medium-high) vulnerability tertiles.

2.4 DISCUSSION

2.4.1 Insights from case study results

Overall vulnerability

The majority of Scotian Shelf species have relatively low vulnerability scores to warming under the mild warming scenario, but 45% of species have relatively high vulnerability scores under the severe warming scenario (+3ºC) given our VPWA model. This could have substantial implications for ecosystem structure, management strategies, and fishing industries in the region. Considering that a near +3ºC bottom temperature anomaly already occurred via an extreme warming event in 2012 (Galbraith *et al*., 2013), these species might be negatively impacted much earlier than 2060, the year for which a +3ºC warming was originally projected.

Many of the species, which obtained high vulnerability scores given our assessment, have recently demonstrated vulnerability to warming events in our region. For example, Snow Crab populations disappeared from the southern part of the Scotian Shelf in 2012 due to record high temperatures in the summer (Jae Choi, DFO, pers. comm.). Further, Capelin has been called the "sea canary for climate change" due to it rapid distributional response to warming (Rose, 2005). This species also displays vulnerability due to its specific habitat requirements in early life stages (e.g., Nakashima, 2002; Penton *et al*., 2012). In these ways, our results seem to agree with available, region-specific literature and local knowledge.

Commercial perspective

At an aggregate level, commercial species have significantly lower vulnerability scores than non-commercial species on average (Figure 2.5). Commercial species are often commercially exploited due to initially high abundance and widespread distribution, reflecting their adaptive nature, and as such, many, but not all, may be least vulnerable to environmental change (Gaston and Fuller, 2008). Of course, commercial species are not indefinitely resilient. Sustainable management of fish stocks will continue to be highly important, especially as overexploitation increases vulnerability to warming (Brander, 2007). For example, annual depletion rates of cod during the 70s-90s throughout the North Atlantic ranged from 46% -67% biomass removal each year for many years, and resulted in a complete collapse of an industry in the northwest Atlantic

(Frank *et al*., 2006). To our knowledge, temperature variability has never had a systematic impact as that caused by these high annual depletion rates. Only diligent monitoring and management can ensure that low vulnerability populations remain resilient, and that stress on highly vulnerable species is not exacerbated (Davies *et al*., 2004; Hutchings and Reynolds, 2004; Brander, 2007; Hsieh *et al*., 2008; Shackell *et al*., 2012; Niiranen *et al*., 2013).

Some key commercial populations are vulnerable under the severe warming scenario: Snow Crab (*Chionoecetes opilio*), WSS Cod (*Gadus morhua*), ESS Pollock (*Pollachius virens*), WSS Pollock (*Pollachius virens*), and WSS Red Hake (*Urophycis chuss*) (Figure 2.4). Industries and communities that are highly dependent on these species/populations could begin developing adaptation strategies to ensure future economic sustainability, and lowered risk to these vulnerable species.

Species At Risk perspective

On average, it appears that overexploited populations may be less vulnerable to warming than Not at Risk and Not Assessed species (Figure 2.6). These populations likely possess other characteristics (other VPWA factors) that will allow them to survive and adapt to environmental changes, including a wide geographic distribution (latitudinal range) and thermal tolerance range (exposure factors). These traits likely contributed to their abundance in the past, but exploitation has reduced their abundance and likely has impacted other population characteristics like life history and size structure (e.g., Hutchings and Reynolds, 2004). For these populations, warming is not the most eminent threat. The survival of these populations (abundance, life history, and size structure), is highly dependent on reduction of exploitive pressures (Shelton *et al*., 2006), especially with the added, though possible minor, stress of climate change (Planque *et al*., 2010). Endangered and Threatened populations are the least vulnerable as an aggregate group. However, some key endangered and threatened species have high vulnerability scores under under the severe warming scenario: WSS Cod (*Gadus morhua*), WSS Cusk (*Brosme brosme*), and Winter Skate (*Leucoraja ocellata*) (Figure 2.4). These species are listed as threatened or endangered because overexploitation has reduced their abundance and recovery potential (COSEWIC, 2012). As a result, these species are less able to adapt to changing conditions, and are more sensitive to additional stress (Planque *et al*., 2010; Perry *et al*., 2010; Brander, 2007). Special care should be taken to protect these species, so that vulnerability to warming is not further exacerbated.

Ecosystem perspective

We found that the most vulnerable species appear to be in lower trophic groups (Figure 2.7); for example, Moustache sculpin (*Triglops murrayi*), Sandlance (*Ammodytes americanus* and *Ammodytes dubius*), and Capelin (*Mallotus villosus*) and have the highest vulnerability scores under the sever warming scenario (Figure 2.4). Sandlance in particular are important prey in the diet of many Scotian Shelf predators (Nelson and Ross, 1991). This could have a dramatic effect on the structure of the Scotian Shelf ecosystem in the near future (e.g., Carpenter *et al*., 1985; Carpenter *et al*., 1987). This is especially concerning as cold-water ecosystems are suggested to be more susceptible to trophic imbalance than warmer-water ecosystems (Frank *et al*., 2007). This could mean that as temperatures increase, warm-water species/populations/ecosystems will benefit and their cold-water counterparts, who are currently occupying the region, will suffer (Fisher *et al*., 2010). The future weakening of an important base to the Scotian Shelf ecosystem could indicate the beginning of an ecosystem-level transformation as warmer-water systems replace native colder-water systems (e.g., Cheung *et al*., 2009).

Species vulnerability is not solely dependent on habitat availability as determined by SDMs (Shackell *et al*., 2014; Cheung *et al*., 2009; Cheung *et al*., in press). Vulnerability (i.e., the risk to a popoulation's future health and abundance) is a function of life history, habitat and diet specificity, range, and projected habitat availability. These factors each impact a population's vulnerability individually and uniquely. That r-selected, mobile species such as Sandlance have obtained such a high vulnerability score is a function of this framework; life history alone does not determine vulnerability.

Importance of scale

Another key finding is that WSS (southern) populations almost always have higher vulnerability scores than ESS (northern) populations (Figure 2.4). This is for both mild and severe warming scenarios. This agrees with a long branch of literature that suggests animals or populations at their more southern range limits (or limits closest to the equator) will be more vulnerable to warming than animals or populations in the more northern (or poleward) stretches of the species' range (e.g., Cheung *et al*., 2009; Cheung *et al*., in press; Shackell *et al*., 2014). This general trend has implications for regional management, as the scale and location of management units will play a large role in prioritizing populations at risk to warming.

Another important use for SDMs and CCVAs is the ability to communicate exceptions to generally accepted trends to managers. Pollock (*Pollachius virens*) is an exception to the increasing trend in vulnerability from ESS to WSS populations. Pollock is currently more abundant on the WSS than on the ESS (Ricard and Shackell, 2013). A large loss of habitat over the entire SS could translate into higher percentages of loss for the less abundant region, the ESS.

2.4.2 Auxiliary Information and the VPWA

Although our VPWA is generalized, provides refined rankings of species/populations, and uses the best information available to incorporate distribution and habitat change projections, there are limitations. It would be prudent to use auxiliary information with the VPWA results. For example, American Lobster has suffered large mortalities in the northeastern USA due to increased incidence of bacterial shell disease, which may also be a consequence of warming (Glenn and Pugh, 2006; Wahle *et al.*, 2009; Tlusty and Metzler, 2012). Due to the uncertainty around the cause of this increasing disease, the difference in location, and that increased disease in lobster is a species-specific case, this could not be considered in our model. Special cases, especially where links to the stressor being evaluated (here warming) are uncertain, should not be included in the assessment, but can be considered as auxiliary information. In our region, which is further north, lobster have not endured the same consequences of warming, likely because the southern limit of this species' range is in US waters; conditions are cooler for this species in our geographic region. That American Lobster did not obtain a high vulnerability score with our VPWA reflects, again, the importance of scale (our evaluation of exposure is specific to our region) and circumstance (the bacteria is established in US waters). However, it should be noted that currently US-based risks could impact Canadian lobsters in the future.

Natural systems change in a non-linear fashion. From a management perspective, it is important to appreciate the complexity of ecosystems. Our VPWA can be used to triage species to identify priorities, and the SDM can be used individually to aid in the adjustment of fisheries management plans (Shackell *et al*., 2014 for details). What these methods do not provide is an understanding of potential ecosystem-level effects on vulnerability, and the effect of population-level vulnerability on ecosystems. We therefore recommend that our framework act as a stepping-stone in the further development of CCVA methodologies, and that future analyses contribute to a broader view of ecosystem-level implications.

Further analyses should also consider other climate drivers, e.g., dissolved oxygen and acidification, as the information becomes available. As our exposure component was highly dependent on climate projections, inclusion of these less-predicted climate drivers was not appropriate. Regional dissolved oxygen and acidification projections are highly uncertain as hisotrical records of dissolved oxygen and pH for the Scotian Shelf bioregion are limited, although the impacts of reduced oxygen on marine animals are well-studied (e.g., Pörtner and Knust, 2007) and some have suggested that acidification may be a larger threat to the global ocean than warming (Turley and Gattuso, 2012). Gaichas *et al*. (2014) included acidification concerns in their community-level CCVA, by assessing risk to populations based on dependence on calcifying organisms and given the current best guess that shellfish and corals are more vulnerable to acidification than teleosts. In our case, this broad dichotomy between telosts and shellfish would not have added any value to our VPWA, as the majority of the species evaluated through the case study are teleosts, and none are, to our knowledge, dependent on calcifying organisms. Further, this topic is highly debated in the literature. A recent paper (Gabay *et al.*, 2014) draws attention to the unresolved nature of acidification debates. While the assumption has been that corals will be impacted by lowered pH, Gabay *et al.* found that colonies of a tropical octocoral (*Ovabunda macrospiculata*) were unaffected when exposed to low pH conditions for 42 days. Further, it has been suggested that teleosts can be negatively impacted by increased CO2, which is the cause of acidification (Pörtner, 2008). However, auxiliary information would be useful to this VPWA where it is well known that further stressors exist (e.g., in the case of shellfish and acidification (Riebesell *et al.*, 2000; Stumpp *et al.*, 2012)).

It is suggested that acidification, warming, and other aspects of climate change may act cumulatively, not individually, on marine species (Hauri *et al.*, 2009; Denman *et al.*, 2011; Gruber, 2011; Doney *et al.*, 2012). Negative impacts of acidification could make some species more vulnerable to warming, and vice versa (e.g., Doney *et al.*, 2012). An ideal CCVA would evaluate the vulnerability of marine species to all cumulative impacts of climate change.

2.4.3 VPWA as another step in CCVA development

Marine animals will react in many different ways to their changing environment. In many cases, the first observable reaction will be a change in distribution (i.e., spatially explicit habitat availability) (Parmesan and Yohe, 2003; Cheung *et al.*, 2009). However, sensitivity will be crucial in determining the future impact of changes in habitat availability on species/population health and abundance. We have incorporated SDM projections into a VPWA that considers sensitivity factors, including adaptive characteristics such as exploitation history (population status) and life history (Musick *et al*. (1999)'s "resilience"). Although various recently developed CCVAs use a similar framework and include many of the same factors (Chin *et al*., 2010; Pecl *et al*., 2011; Reece *et al*., 2013; Gaichas *et al*., 2014), ours builds on the standard approach to improve utility for managers in regional-scale climate adaptation programs. First, our VPWA defines exposure as a function of gain/loss of thermal habitat at multiple life stages. This is an important step in CCVA development as earlier life stages are known to often have a "bottlenecking" effect on a species overall vulnerability to environmental change (Cushing, 1997; Pörtner and Farrell, 2008; Mantzouni and MacKenzie, 2010; Petitgas *et al*., 2013). Second, our results show that geographical location and scale is highly important to the outcome. As ocean managers tend to plan at small and variable spatial scales, it is important that adaptation tools provide information specific to their management area. Finally, our numerical scoring approach allows for more refined ranking of species. Improvement of quantitative approaches was identified as an important next step in CCVA development by the IPCC in 2001. The scoring method makes it easy to quantify qualitative knowledge from the literature. Refined triaging of populations by their vulnerability will make it easier for marine managers to identify priorities from the model results.

The flexibility of this VPWA allows for its modification in other regions. This VPWA can be used to assess any species/population for which distribution, temperature tolerance, life history, population, habitat, and diet information exist. This assessment has well-founded biological and ecological underpinnings and is useful for marine managers working in a time of climatic uncertainty. For this reason, the VPWA is broadly relevant as a starting point. However, factors will change depending on data availability in other regions where the tool may be applied. For example, thermal window analyses may be used in lieu of a SDM in regions where species distribution data are limited, and an exposure factor for sea ice loss should be included when the tool is applied in polar regions.

2.5 CONCLUSION

Unless extreme measures are taken to reduce carbon emissions, it is nearly impossible to reverse the trajectory of global climate change (IPCC, 2013). Physical and chemical changes will proceed in the ocean, with biological changes to follow. As warned by the extreme summer temperatures in 2012 on the Scotian Shelf, these changes may occur sooner than physical oceanographers have projected. Consequently, the need for tools to inform climate adaptation programs is increasingly urgent.

We have made steps toward a broadly applicable semi-quantitative VPWA, but recognize that this methodology can be improved. We were only able to consider warming due to uncertain forecasts of, and unknown responses to, acidification and dissolved oxygen. As more information becomes available, estimates of future pH and oxygen, coupled with an understanding of how these changes will impact marine animals, will be appropriate additions to this framework.

Nonetheless, our VPWA uses the best information available to assess and rank regional populations by their vulnerability to warming, a significant and likely climate shift. This will provide an initial base of information upon which managers can identify priorities as well as needs for supplementary information and research. This VPWA can be helpful for regional climate adaptation programs now.

CHAPTER 3 AVENUES FOR INCLUDING THE FULL SCOPE OF CLIMATE DRIVERS IN MARINE CLIMATE ADAPTATION TOOLS

3.1 INTRODUCTION

Climate change, caused by increased emissions of CO_2 and other greenhouse gases to the atmosphere, affects major ocean forces, including warming, acidification, ice melt, decreased salinity, stratification, and oxygen depletion. The effects of all climate change stressors are likely to be synergistic in that many may exacerbate the effects of the others (Pörtner, 2008; Gruber, 2011; Doney *et al.*, 2012). Warming can directly cause an increased metabolic rate in animals (Denman *et al.*, 2011). Often, marine populations will respond to warming by shifting in distribution, which can cause ecosystem-level changes (Cheung *et al.*, 2011; Shackell *et al.*, 2014). Warming can also increase the occurrence of hypoxia in shallow, enclosed areas and deep waters due to the fact that oxygen dissolves less readily in warm water than in cold water, and oxygenated surface waters may not reach greater depths when stratification occurs. Exacerbating this issue, increased CO_2 in the ocean can cause hypercapnia in the internal fluids of marine animals (Pörtner, 2008). Increased use of oxygen due to increased metabolism in combination with decrease oxygen availability and increased CO_2 in the internal fluids can cause major physiological problems for marine species that are unable to acclimate (Pörtner, 2008: Denman *et al.*, 2011; Gruber, 2011). Further, reductions in sea ice cover is causing accelerated warming due to reduced albedo, and large inputs of fresh water from melt. It is hypothesized that the combination of accelerated warming and decreasing surface salinity in polar regions may cause increased stratification, which could cause global ocean circulation to slow or stop (Mitchell *et al.*, 2006; Alley, 2007; Dima and Lohmann, 2010). Increased storm events are also a source of decreased salinity in ocean surface waters in other regions (Mitchell *et al.*, 2006). If these changes progress as expected, future ocean ecosystems could differ dramatically from those of today.

With such potentially dramatic changes pending, global interest in climate adaptation tools has risen (e.g., DFO, 2013b), as discussed in previous chapters. Species Distribution Models (SDMs; e.g., Shackell *et al.*, 2014) and Climate Change Vulnerability Assessments (CCVAs; e.g., Chapter 2) are promising tools; however, neither has, as of yet, been designed to quantify the effects of climate change stressors other than temperature on the distribution and vulnerability of marine species. Nor has either explored the cumulative impacts of these stressors on marine life. In this chapter, I discuss how climate stressors other than warming have been considered in climate adaptation tools to date. I then discuss the data that is currently available (at both global and regional scales), where more data are needed, and possible avenues for improvement of climate adaptation tools while data is limited.

3.2 CLIMATE STRESSORS IN ADAPTATION TOOLS TO DATE

Until recently, species- and community-level CCVAs have been highly descriptive, likely due to a lack of high-resolution climate projections and/or species distribution data. These VAs have used the information available to qualitatively estimate the exposure, sensitivity, and adaptive capacity of marine species or communities to relevant climate change factors, i.e., warming, acidification, hypoxia, sea ice melt (e.g., Laidre *et al.*, 2008; Moore and Huntington, 2008; Chin *et al.*, 2010; Mamauag *et al.*, 2013; Gaichas *et al.*, 2014). Recently, Gaichas *et al.* (2014) qualitatively ranked fish community vulnerability to acidification, oxygen depletion, stratification, increased winds and storm events, and sea-level rise (those climate stressors deemed significant in the Gulf of Maine and Mid Atlantic Bight regions). Ranks for community exposure and sensitivity to these stressors were derived from literature and expert opinion; rank was based on the probability that negative impacts would occur and the varying intensities of impacts anticipated for different habitat types. This approach was adapted from an earlier species-level CCVA (Chin *et al.*, 2010). However, no community or species-level CCVAs have, as of yet, quantified species exposure to these and/or other climate stressors such as melting sea ice (relative to region) over time. As I will discuss in the following sections, appropriate data for quantitative methods, like the inclusion of SDMs, are limited in the majority of regions. The VA designed in this thesis could only consider warming because of a lack of spatial data for other climate stressors. Further, the effects of these stressors on marine biological systems are not completely understood.

Cheung *et al.* (2011) projected changes in the position of marine populations' centroids using generalized projections for temperature, oxygen concentration, and pH at the global scale. These projections do not include variability with depth and location; therefore Cheung *et al.*'s projections are highly generalized. They do provide a first-level glimpse at what the biophysical impacts of climate change could be for many marine populations, globally. Shackell *et al.* (2014) developed an SDM, which they used to project marine species realized thermal habitat at a higher resolution, under downscaled warming scenarios for the Scotian Shelf and Northeastern USA. These warming scenarios also roughly accounted for variability with depth. In Chapter 2 of this thesis, I incorporated the results of Shackell *et al*. (2014) into a species-level CCVA for the region. Projections of habitat gain or loss determined how species were ranked in terms of "exposure" to climate change. However, loss of habitat as a result of acidification, decreasing salinity, stratification, sea level rise, increased storm events, or oxygen depletion is not projected for any time frame. In the case of the Scotian Shelf, this was due to a lack physical and chemical oceanographic data and projections at that time. On the Scotian Shelf, salinity and stratification data and projections are available (Loder *et al*., 2013), but these factors threaten eggs and larval stages of animals (Chabot *et al*., 2013), for which there are limited spatial and biophysical tolerance data (Shackell and Frank, 2000). As more data become available, or as researchers become able to project pH and oxygen given other monitored chemical components in the ocean, the predictive ability of SDMs can be improved. This is important because temperature is not the only environmental variable impacting marine species "fundamental" niche (Wake *et al*., 2009), i.e., where their preferred environmental conditions exist.

If species preferences for all conditions, temperature, pH, etc., were known, and spatial data for all these conditions were available, cumulative exposure of the fundamental niche could be measured using an SDM. This fundamental niche is further constrained by species interactions (predator/prey relationships and competition). The constrained niche is called the "realized" niche (Wake *et al*., 2009). It is very difficult to project the realized niche using spatially-explicit methods, so high-resolution SDMs have not tackled this issue as of yet, though this is one of the identified limitations of these tools (Pearson and Dawson, 2003).

Until data availability improves, regional-scale SDMs should use the best data available (e.g., Shackell *et al.*, 2014). Our VPWA model can be improved to qualitatively consider other climate change stressors where they are relevant. However, qualitative consideration of these stressors would require an understanding of their potential impacts in relation to species exposure and sensitivity: what life stage(s) will be impacted, what traits make a species more or less sensitive, and what habitat types will be most impacted (e.g., Chin *et al.*, 2014; Helmuth *et al.*, 2014). These topics are widely debated and information is limited, but available information should be used to provide context for VPWA results where appropriate.

3.3 PHYSIOLOGICAL AND PHYSICAL EFFECTS OF CLIMATE CHANGE

Below is a review of what is known with respect to the impacts of various climate stressors on marine species in general. That each stressor is likely to impact different regions and different species in diverse ways is discussed briefly also.

3.3.1 Warming and oxygen depletion

All marine organisms have a range of temperatures within which they can survive (Pörtner and Peck, 2010). With warming, some marine organisms may be forced out of their current habitats due to temperatures extending outside this range. Physiologically, animals experiencing temperatures outside their tolerance range can have an increased metabolic rate, and reduced growth and reproductive capacities (Pörtner and Peck, 2010). Branco *et al.* (2013) demonstrated that some urchins experience a decreased immune response under warming conditions. Pimentel *et al.* (2012) found that squid and cuttlefish, when exposed to warming, experience a rapid increase in oxygen consumption, metabolic suppression toward the end of embryogenesis, and oxygen depletion in egg capsules. They speculated that, in such conditions, younger life stages may require greater amounts of food to compensate (Pimentel *et al*., 2012). Repolho *et al*. (2014) found that even a 3-degree increase in temperature shortens embryonic development time, decreases survival, decreases size at hatch, increases the number of premature larvae, and increases metabolic rate in octopus.

Invertebrates are physiologically more sensitivity to such changes than vertebrates due to highly sensitive metabolisms and reduced ability to maintain internal homeostasis (Pörtner, 2008). However, similar effects have been observed in fish populations, such as decreases in body size with warming (e.g. Baudron *et al*., 2014). These physiological effects can reduce the viability of populations and cause shifts within the ecosystem, especially when other stressors such as habitat destruction and over-fishing are at play. Current trends point to a decrease in body sizes for many marine animals as a result of the combined physiochemical effects of warming, oxygen depletion, and exploitation (Bundy and Fanning, 2005; Cheung *et al*., 2013; Baudron *et al*., 2014). Sustainable fishing practices will be crucial in sustaining resilient marine populations in such heightened stress contexts.

In the case of some photosynthesizing species, significant warming can be lethal. With some zooxanthellae (*Symbiodinium* spp.), lethal warming means these species can no longer provide nutrients to symbiotic corals (Van Oppen *et al*., 2005; Van Oppen and Lough, 2009; Hoegh-Guldberg and Bruno, 2010). This phenomenon, called "coral bleaching", has been observed across the globe and can result in the death of entire coral reefs, which are home to and feeding grounds for a large array of marine life (Birkeland, 1997; Van Oppen and Lough, 2009). Loss of coral reefs to bleaching can be detrimental to the diversity and health of ocean ecosystems.

Some individuals and populations can avoid unfavourable conditions, thereby preventing negative physiological effects, by migrating or dispersing to new habitats. Species distribution shifts can be projected using SDMs if temperature tolerance windows are known; however, as previously discussed, temperature is only one factor affecting species distribution. Poleward migrations of fish and invertebrates are being observed as warm water species are invading previously cooler regions (Pinsky *et al.*, 2013). Some warm water populations are projected to expand in spatial distribution and increase in abundance as a result of warming (increased habitat availability) and/or resulting decreases in competition (Cheung *et al.*, 2013; Gilly *et al.*, 2013; Kjesbu *et al.*, 2014). Observations in the Barents Sea indicate that an increase in spawning stock biomass for a local population of Atlantic cod (*Gadus morhua*) occurred as a result of warming and decreased fishing pressure (Kjesbu *et al.*, 2014). Warming has expanded the area available for cod to feed, thereby creating greater habitat carrying capacity for the population. Results of ecosystem models (i.e., food web models to predict changes in species interactions) have demonstrated that decreased fishing pressure and decreased nutrient loading could also contribute to increased abundance of commercially important fish species in northern waters (Niiranen *et al.*, 2013). Atlantic herring and cod stocks are projected to increase with gradual warming in combination with sustainable fishing practices (Niiranen *et al.*, 2013; Bartolino *et al.*, 2014; Kjesbu *et al.*, 2014).

On the other hand, oxygen does not dissolve as readily in warm water as it does in cold water (EPA, 2012). As warming causes a shoaling of the Oxygen Minimum Zone (OMZ), some habitats may no longer be suitable for oxygen-dependent, water-breathing animals (e.g. Gruber, 2011; Gilly *et al.*, 2013). This will be a larger issue in enclosed ecosystems (bays, estuaries, deep basins) than in the open ocean (Gruber, 2011). In fact, rapid oxygen depletion was observed in the St. Lawrence estuary and the Gulf of St. Lawrence between the 1930s and 1980s (Gilbert *et al.*, 2005). Since then, oxygen levels have been fairly stable, but oxygen levels may decrease again with warming. Further, increased influx of organic matter, and increased respiration due to accelerated metabolisms may have exacerbated the effects of decreased oxygen in the past, and could again in the future.

In areas such as the Gulf of St Lawrence, the St. Lawrence estuary, and some deep basins ecosystems may shift dramatically as a result of warming and oxygen depletion. It has been suggested that highly mobile animals will be at highest risk because lowered oxygen conditions do not support high metabolisms (Childress, 1995; Hochachka and Somero, 2002; Seibel and Drazen, 2007). Exacerbating this issue, warming accelerates animals' metabolisms. Thus, small, slow-moving animals may replace large, fast-moving animals in warming, oxygen-depleted regions (e.g. Niiranen *et al.*, 2013). It has also been suggested that large, sedentary, benthic species are highly susceptible due to decreasing oxygen in deep habitats (Clark *et al.*, 2013). In addition to possible decreases in body size of fish and dramatic changes in the deep, warming may cause larger phytoplankton to be replaced by picoplankton, which will likely have bottom-up impacts on ecosystems (Daufresne *et al*., 2009; Finkel *et al*., 2009; Suikkanen *et al*., 2013).

3.3.2 Acidification

Increased CO_2 in the atmosphere causes increased dissolution of CO_2 into the ocean. First, overabundant CO_2 reacts with carbonate (CO_3^{2-}) to produce bicarbonate (HCO_3^-), thereby reducing the concentration of calcium carbonate ($CaCO_3$) available to shell-building animals (e.g. Gruber, 2011). Second, an overabundance of CO_2 can cause acidosis in the internal fluids of some other marine animals. Acidosis, increased acidity in the blood or other tissues, can cause severe to lethal damage (Frommel *et al*., 2012). Additionally, avoidance of acidosis through up-regulation of processes contributing to homeostasis can interfere with neurotransmission in fish (Nilsson *et al*., 2012). In the worst of cases, this can cause fish to be attracted to things they normally are not, their predators for example (Dixson *et al*., 2010). It has also been observed that increased CO_2 levels can enhance the sensitivity of organisms to warming as a result of reduced tissue function and visa versa (Pörtner *et al*., 2005; Metzger *et al*., 2007). On the other hand, an increase in CO_2 in the ocean benefits photosynthetic organisms, which require CO_2 for photosynthesis (Hendriks *et al*., 2010). In enclosed areas, a dramatic increase in phytoplankton abundance at the surface can lead to eutrophication and oxygen depletion at greater depths (Rabalais *et al*., 2009; Moss *et al*., 2011). This can directly impact heterotrophic animals, which require oxygen for respiration.

In lower invertebrates, especially shell-building organisms, the effects of acidification will be most significant, in most cases; performance, reproduction, behaviour, and growth may be permanently impaired (Pörtner, 2008; Hendriks *et al.*, 2010). Decreased availability of calcium carbonate in the ocean increases the metabolic cost of calcification in shell-building organisms (Stumpp *et al.*, 2012). This can lead to increased mortality, especially in early life stages when shell growth is crucial (Orr *et al.*, 2005; Stumpp *et al.*, 2012; Ceballos-Osuna *et al.*, 2013). It is projected that the ocean will experience a 25% decline in biological calcification within the 21st century if CO_2 emissions are not significantly reduced (IPCC, 2013). Ocean acidification events of the magnitude expected for the year 2100 have caused mass extinction events in our geological past (Pörtner *et al.*, 2004; Knoll *et al.*, 2007). However, some calcifying organisms may be less vulnerable than originally thought (e.g., Gabay *et al.*, 2014 on unexpectedly low sensitivity of an octocoral species to acidified conditions).

Some calcifying organisms play very important roles in ocean ecosystems and human economies; therefore acidification could have cascading impacts. Pteropods are eaten by many other marine animals in Arctic ecosystems (Orr *et al.*, 2005), while corals, those that may be vulnerable, provide shelter and feeding grounds in the tropics and temperate regions (Birkeland, 1997; Van Oppen and Lough, 2009; Baker *et al.*, 2012). Tropical corals are also at high risk due to bleaching. Shellfish filter water and feed on fallen particles; this is an important role, which they fill in every marine ecosystem on the planet. Further, shellfish are an important source of protein for human populations worldwide. Identifying dependencies of marine animals on calcifying organisms can be a way to include acidification and species interactions into a CCVA (e.g., Gaichas *et al.*, 2014).

3.3.3 Sea Ice and Salinity

Increasing rates of sea ice melting in polar and sub-polar regions are having cascading effects in the global ocean. First, melting ice creates a positive feedback loop for further warming and melting due to decreasing albedo effect (e.g., Kellog, 1975; Gruber, 2011). Second, melting ice increases fresh water input to the ocean, which can increase sea level and stratification (Gruber, 2011). Increased stratification can reduce mixing, which can lower transport of oxygen and nutrients to benthic ecosystems and, perhaps, slow overall ocean circulation (Mitchell *et al*., 2006; Alley, 2007). Further, reduced sea ice cover in polar regions reduces important breeding, nursing, feeding, and travelling habitat for seals, walrus, beluga whales, bowhead whales, polar bears, and narwhals (Simmonds and Isaac, 2007; Kovacs and Lydersen, 2008; Moore and Huntington, 2008; Hunter *et al*., 2010). Ecosystems below sea ice are unique and often highly diverse with species specialized for such conditions (Post *et al*., 2013).

In other regions, salinity is changing due to shifts in the precipitation/evaporation balance. Increased precipitation and run-off, in combination with contributions from melting sea ice, in some regions is resulting in lower surface salinity (Gruber, 2011; IPCC, 2013). In other regions, reduced precipitation and increased temperatures are resulting in increased evaporation of water, which leads to highly saline surface waters (IPCC, 2013). In either case, the surface pelagic ecosystem will likely be most impacted. Where surface salinity decreases, egg and larval stages of marine animals could lose buoyancy (Chabot *et al*., 2013). Where surface salinity increases, animals' internal chemistry could be impacted. However, little is known about the tolerance of marine animals to changes in salinity (Chabot *et al*., 2013). What is known suggests that salinity is not as significant a stressor as warming, acidification, melting ice, or decreased oxygen.

3.3.4 Sea level rise

Sea level rise will likely be an issue where marine turtles and seabirds nest and/or feed on beaches or rocky shores (Galbraith *et al*., 2002; Baker *et al*., 2006; Poloczanska *et al*., 2009; Fuentes *et al*., 2010; Seavey *et al*., 2011). Sea level rising over important nesting beaches/shores could leave only developed land beyond the water line. Significant reductions in nesting habitat could have severe consequences for population viability (Galbraith *et al*., 2002; Baker *et al*., 2006; Poloczanska *et al*., 2009; Fuentes *et al*., 2010; Seavey *et al*., 2011). Sea level rise will also be an issue for human coastal communities; at-risk harbours, houses and other such structures may need to be relocated over time (Shackell *et al*., 2013). Additionally, beach, rocky shore, and mangrove ecosystems can provide coastal regions with protection from erosion and storm surges (Titus, 1986; Ellison and Stoddart, 1991; Thampanya *et al*., 2006; Hawkins *et al*., 2008). If sea level reaches developed land, protection from erosion will not exist and coastal communities will be more at risk of damage from natural forces (Shackell *et al*., 2013).

3.3.5 Synergistic Effects

Warming, oxygen depletion, acidification, sea ice melt, and decreased salinity will each exacerbate the impacts of one another (Pörtner, 2008; Denman *et al*., 2011; Gruber, 2011; Doney *et al*., 2012). Consequently, these stressors will have synergistic impacts on marine species and ecosystems. For example, warm water cannot hold as much oxygen as cold water, but will hold more CO_2 (Gräns *et al*., 2014; Keeling *et al*., 2010; Moss *et al*., 2011; Seibel, 2011; Suikkanen *et al*., 2013), which may exacerbate the problem of acidification. The oxygen minimum zone and carbonate saturation horizons are shoaling as oxygen levels decrease and carbon dioxide levels increase at depths (Hauri *et al*., 2009; Seibel, 2011). Additionally, surface waters at higher latitudes are already more acidic than the rest of the ocean, and tropical waters are already lower in oxygen. Decreased ocean mixing due to warming and increased fresh water input may exacerbate this issue, as oxygen-rich surface waters may not reach the depths (Mitchell *et al*., 2006; Alley, 2007; Dima and Lohmann, 2010). Further, increased phytoplankton abundance in some areas as a result of increased CO_2 may increase the quantity of decomposing material falling to the bottom, causing further reductions in oxygen. As a result of these synergistic effects, hypoxia-intolerant and shell-building species will likely loose habitat in deeper areas (Gruber, 2011). Warming and freshening at the surface, and oxygen depletion and acidification in the deep will make it difficult for many animals to find suitable habitat in the future ocean. Further, species-interactions could have unpredictable effects on the future state of climate change-impacted marine ecosystems. As aforementioned, a note of caution is that the conditions expected for the year 2100 caused major extinctions in our geological past (Pörtner *et al*., 2004; Knoll *et al*., 2007).

Literature stressing the importance of considering cumulative impacts in risk assessments is plentiful (e.g., Harley *et al*., 2006; Stelzenmüller *et al*., 2010). However, advice on how to quantitatively predict synergistic effects for vulnerability assessments is vague or suggests that impacts are additive (Cocklin *et al*., 1992; Teck *et al*., 2010; DFO, 2013b). Cocklin *et al*. (1992) suggested that evaluating the current state of systems affected by multiple stressor could help guide future predictions of cumulative impacts. However, this has not yet been done. This is likely due to the fact that the magnitude of individual and synergistic impacts will differ among species and among stressors (Gruber, 2011); a standard method cannot be applied to all species or stressors. Further, cumulative impacts may change in unpredictable ways over time (not necessarily in a linear fashion), especially if systems are already at a threshold when their initial state is assessed (Cocklin *et al*., 1992; Dubé, 2003). As a result, it is common practice in vulnerability assessments to consider multiple stressors separately and evaluate overall impact as the sum of their individual, equally weighted impacts (e.g., Chin *et al*., 2010; Gaichas *et al*., 2014).

3.4 AVAILABLE DATA AND PROJECTIONS

The Intergovernmental Panel on Climate Change (IPCC) has provided global projections of warming given various CO_2 emissions and assimilation scenarios (IPCC, 2013). The majority of regional-scale projections are downscaled from the IPCC's "status quo" scenario. This scenario follows current climate trends and projects into the future given the assumption that carbon emissions will not be significantly reduced from current levels. Unfortunately, the resolution of downscaled climate projections is low. Consequently, scientists cannot make accurate estimations concerning ocean temperature at regional scales and at differing depths (Loder *et al*., 2013). Projections concerning the future of oxygen, salinity, and carbonate in the ocean are even less certain and more generalized. It has been projected that the entire ocean will be undersaturated with respect to carbonate by the year 2100 (Orr *et al*., 2005). Also, given that global sea and land ice is melting at a rate of or 2.8 mm/yr (IPCC, 2013), it is projected that all glaciers will be reduced by 35-85% by the end of the 21st century.

On the Scotian Shelf, it is predicted that, in 50 years, sea surface temperature will increase by 3-4ºC, pH will decrease by 0.1-0.2 units, the deep basins may experience reductions in oxygen of 30-40%, surface salinity will decrease by 0.1-0.3 units (Chabot and Gilbert, 2013). From these projections, it is evident that warming, acidification, and decreasing oxygen will be significant climate stressors in the Scotian Shelf marine bioregion over the next 50 years. Warming projections are improving in resolution, but oxygen projections are in the beginning stages, and pH data are limited both spatially and temporally. At present time, warming is only projected for the surface. Projections for all climate stressors are less certain below the surface, but the general consensus is that the effects of warming and freshening decrease with depth (Chabot and Gilbert, 2013; Hebert, 2013), and that the effects of acidification and hypoxia will increase with depth (Hebert, 2013). Although oxygen is not monitored sufficiently on the Scotian Shelf, survey data for the Gulf of St. Lawrence include records of oxygen concentration from 2004-2013. In this region, oxygen depletion will likely be a significant threat in the future as it has been in the past (Gilbert *et al.*, 2005). Small increases in stratification have also been observed since 1979 throughout the Northwest Atlantic Ocean (Loder *et al.*, 2013). It has been noted that higher resolution data is needed to improve the resolution of projections for the Northwest Atlantic (Loder *et al.*, 2013). Without detailed projections for all climate change stressors at regional scales, species distribution shifts and/or vulnerability cannot be projected with accuracy. For example, animals in some deep basins on the Scotian Shelf may not be exposed to significant warming, but may experience loss of habitat due to lowered oxygen levels, or, in the case of shell-building organisms, shoaling of carbonate saturation horizons.

3.5 AVENUES FOR IMPROVEMENT OF CLIMATE ADAPTATION TOOLS

At the global scale, climate change projections are highly generalized, but can be used in first-level risk, vulnerability and/or range shift analyses. Although global scale marine SDMs, like that of Cheung *et al.* (2011), can estimate global marine biodiversity and species richness over the long term, marine managers work at regional scales and require regional-scale, short-term forecasts. Given spatial data availability and the low resolution of regional-scale climate projections, warming may be the only climate driver that researchers can include in SDMs in many regions at the present time. However, as more data become available, it will be necessary to include other climate change stressors in these models.

The VPWA model developed in this thesis is ideal when robust SDMs are available. It relies on high-resolution spatial data to make it more relevant to regional marine managers. Its semi-quantitative nature allows for a refined triaging of species, which can inform managers' prioritizations of resources over the short term. However, it cannot provide a complete picture of species' vulnerability to climate change without considering species exposure to all climate change stressors. Ideally, data availability would improve so that more accurate SDMs could be included in this semi-quantitative VPWA. Until more data become available, a next step could be to include qualitative assessments of species exposure and sensitivity to climate drivers without spatial data.

Gaichas *et al*. (2014) considered indirect impacts of acidification in a CCVA, which evaluated community-level vulnerability for fish stocks adjacent to the northeastern USA. Fish communities that depend largely on calcifying organisms for food or shelter were considered more sensitive to acidification than those that did not depend on such organisms at all. Chin *et al*. (2010) qualitatively evaluated shark and ray species exposure to climate change stressors, including changes in ocean mixing, according to their habitat types and dependencies; e.g., species dependent on coral reef ecosystems were deemed more exposed to acidification impacts than pelagic species. Gaichas *et al*. (2014) explained that their methods for evaluating community exposure were similar to those of Chin *et al*. (2010). Similar methods have also been used to describe Arctic marine mammal vulnerability to climate changes, including sea ice melting (e.g., Laidre *et al*., 2008; Moore and Huntington, 2008). This approach can rank species by their vulnerability to climate change, considering all relevant climate drivers, but ranking is coarse and there is no real temporal or spatial component, nor are they based upon spatially explicit climate projections. Further, there is much uncertainty concerning the relative sensitivities of different animal groups to these other climate drivers (e.g., Gabay *et al*., 2014). Such coarse, uncertain ranking may not be as relevant to regional fisheries management programs. Until debates are settled concerning relative sensitivities, and more data and projections become available, this information may not be appropriate for VA models.

3.6 DISCUSSION

Ideally, detailed projections would exist for all climate change stressors discussed, tolerance levels for these stressors would be known for a wider range of species, and indirect consequences through species interactions would be better understood. If this were the case, the results of SDMs and semi-quantitative CCVAs would more accurately resemble the future distribution and vulnerability of marine populations. In order to reach this goal, seasonal oceanographic surveys should record oxygen concentration, pH (and/or carbonate concentrations), and salinity with each tow; improved temporal and 3-D spatial coverage of oxygen, pH, and salinity records are needed. These records can be used to observe trends over time and then to build projections for the future. These trends and projections should be developed at regional scales and should include variation with depth. Survey data for marine populations should have corresponding oxygen, salinity, and carbonate records so that SDMs can be applied.

SDMs and CCVAs can be used to estimate the impact of climate change on individual species health and their "fundamental" niche. However, species predator/prey relationships and competition narrow the fundamental niche and limit population growth. A combination of climate change stressors and species interactions will determine species future "realized" niche (Wake *et al.*, 2009). For this reason, future studies should attempt to use the results of SDMs and CCVAs to model future ecosystem-level changes. Ecosystem modeling tools such as EcoSimR (Gotelli and Ellison, 2013) could facilitate this step of analysis. This would improve our understanding of how distribution shifts and high vulnerability of some species may impact others. Further, this could help in the development of species interaction factors within a fully "realized" CCVA.

Although there are many ways in which SDMs and CCVAs can be improved, they can, in their present form, provide useful information for regional and/or international climate adaptation and marine conservation programs. Further, these tools represent an evolving art; each step in the evolution is important. Adaptation tools will use the best information currently available to help human communities make informed decisions for the future. The methods used should depend on the data available as well as the intended audience (i.e., regional, international, species conservation, fisheries management).

CHAPTER 4 CONCLUSION

Although the ideal or fully “realized” CCVA may not be possible in the near future due to data limitations, the VPWA designed in this thesis represents an important step forward. It represents a successful combination of VA and SDM methods, and subsequently improves on quantitative approaches to CCVAs. In this way, this work has achieved the original objective. This thesis has also provided a partial answer to the question, how do we govern for climate change under uncertainty. CCVAs and SDMs are important tools for governing under uncertainty as they can identify priorities/ areas of concern for marine managers. The VPWA designed in this thesis makes use of the best information currently available to do just that, and makes the output specifically relevant to managers at regional scales. However, governance for climate change in the ocean will require additional actions following the identification of vulnerable species/populations. The identification of potential priority species/populations should initiate further monitoring and research on the impacts of climate change on these species/populations, including climate drivers other than warming. Additionally, in the case of highly vulnerable commercial species, the appropriate climate adaptation response from fisheries management divisions may be to consider warming in stock assessments for these species/populations, so as to limit risk of overexploitation in years when warming is a greater threat.

In Canada, this VPWA is an important contribution to the Aquatic Climate Change Adaptation Services Program (ACCASP). This framework can be adapted in other bioregions to identify priorities. On the Scotian Shelf, results of this VPWA have some crucial implications. Although short-term (mild) projections do not lead to high vulnerabilities in many key local species, long-term projections do. Temperatures projected for the long term (2060/severe) have occurred in the area during an extreme warming event. This implies that managers may wish to base priorities for the not-so-distant future on results given for the severe warming scenario. Results offer insight into how warming may impact various management divisions, and which species/populations may require additional monitoring and management.

This VPWA is flexible enough to allow for its adaptation in other regions where similar data exist, and methods are easily repeatable. However, caution should be taken when adapting this and other CCVAs or SDMs for other regions, as the subjective nature of this tool requires that every step in its manipulation be extensively justified. Sensitivity analysis is a useful method for testing the utility of included factors and the effects of their weighting. It is also important to ensure that all life stages, and all key life history, habitat, diet, and population characteristics are considered; there are many factors that determine species vulnerability. This thesis may be used as a guidebook for researchers intending to continue CCVA development, or to use the framework for other purposes.

There are many ways to design and conduct vulnerability assessments. For this reason, the method should be chosen with data availability and the intended audience in mind. The VPWA designed in this thesis used the best information available at a regional scale to ensure applicability for marine managers. As some of the analyses used, and some of the factors included, were a function of the data available in the case study region, methods may be adapted for use in other regions where data availability differs. For example, thermal window analyses may be used in lieu of SDMs where species distribution data are limited. Further, that warming was the only climate driver considered in this model was a function of data availability and relevance to the case study region; when the tool is adapted for use in other regions, consideration of other climate drivers may be necessary (e.g., sea ice melt in polar regions).

Climate adaptation remains a very new science. In the infancy of this field, a combination of quantitative and qualitative methods, and use of auxiliary information to provide context, may be the best way forward while data remain limited. However the semi-quantitative VPWA framework designed here can and should be built on as more data become available concerning projections and biological impacts of other climate drivers, and as our understanding of ecosystem-level effects improves.

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APPENDICES

APPENDIX A: THE ORIGINS OF THE PROJECT

This work came out of discussions with various marine management officials planning for climate adaptation in MPA planning, species at risk management, and fisheries management across Canada, as part of an Aquatic Climate Change Adaptation Services Program (ACCASP) (DFO, 2013b). They requested help on how to allocate resources: do they have to worry about warming? Will it affect how they plan MPAs? Will all species move out of the area, deeming MPAs useless? Which fisheries might be affected? How can we adjust our quotas for climate change?

The SDM developed by Shackell et al. (2014) will answer some questions for fisheries managers and the work suggests some ways climate change can be integrated into the fisheries stock assessment process. Knowledge of the rate at which populations may shift in distribution can help fisheries make decisions about reallocation of resources. The SDM also answers the question, "will all species move out of the area, deeming MPAs useless?" Indeed, not all species will move out of the region in the next 50 years.

Our VPWA furthers the answers to these questions. Our VPWA triages species to help managers identify priority populations, reiterates that not all species will be very negatively impacted by warming, and suggests that MPAs built on the concept of "enduring features", including common, less vulnerable species, will be highly useful.

These two projects have contributed useful adaptation tools to the ACCASP program. These tools help to improve the way that scientific information is translated into management decisions.

References

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APPENDIX B: SDM RESULTS FOR WSS AND ESS POPULATIONS OF SEVEN CASE STUDY SPECIES

These results were obtained via the SDM built in Shackell *et al.*, 2014.

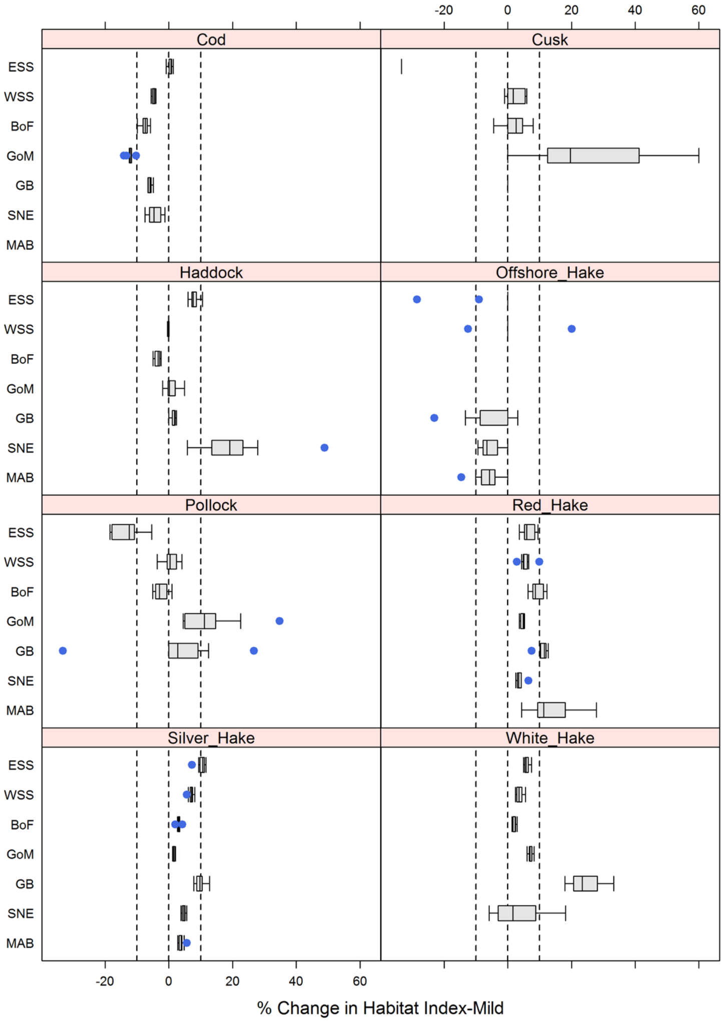


Figure B.1. These boxplots represent habitat change from SDM habitat model from normal (now) to mild warming scenario from 10 model fits using subsampled data for each Gadoid species. Regions are defined in Shackell *et al*., 2012: ESS = eastern Scotian Shelf, WSS = Western Scotian shelf, BoF = Bay of Fundy, GoM = Gulf of Maine, GB = Georges Bank, SNE = Southern New England, and MAB = Mid-Atlantic Bight. Various estimates for region/species combinations are missing indicating a poor model fit (Shackell *et al*., 2014)

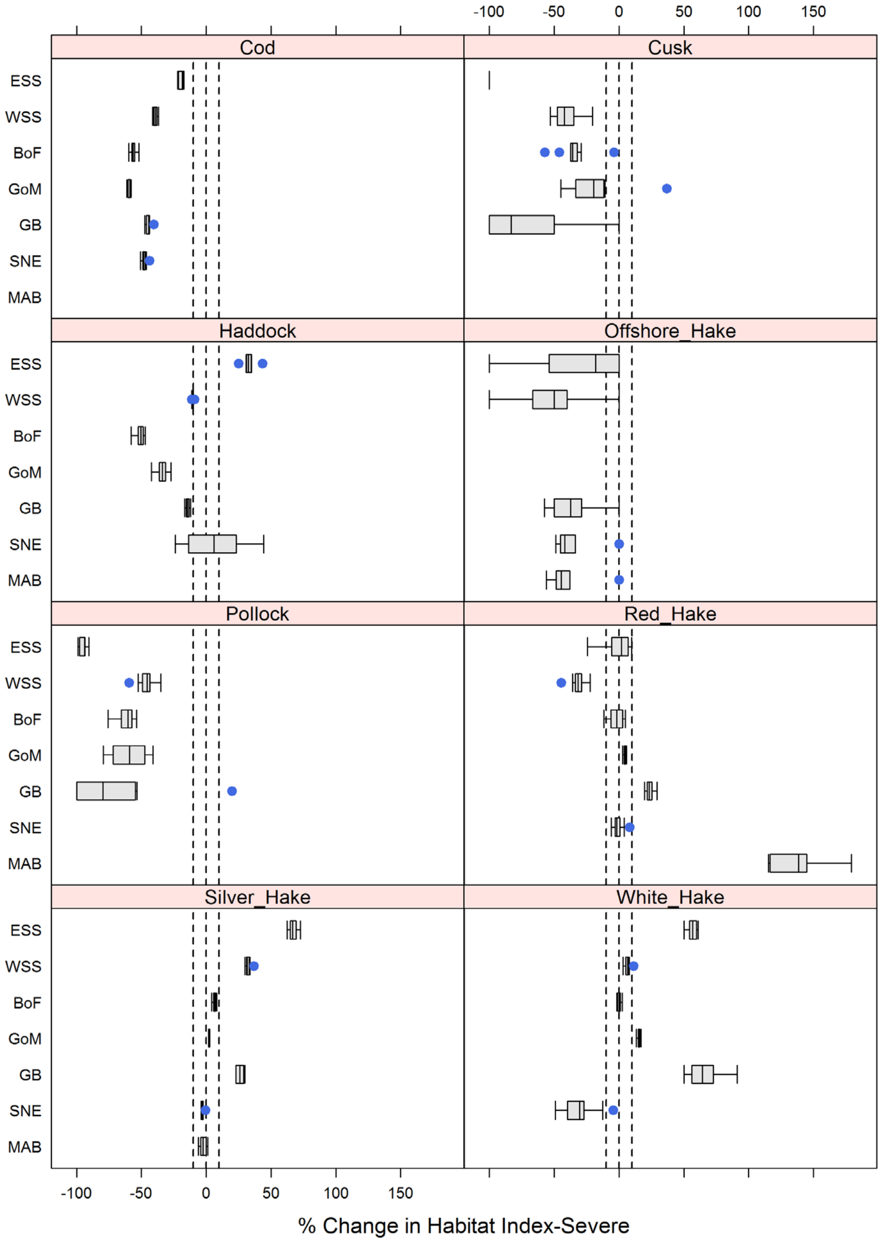
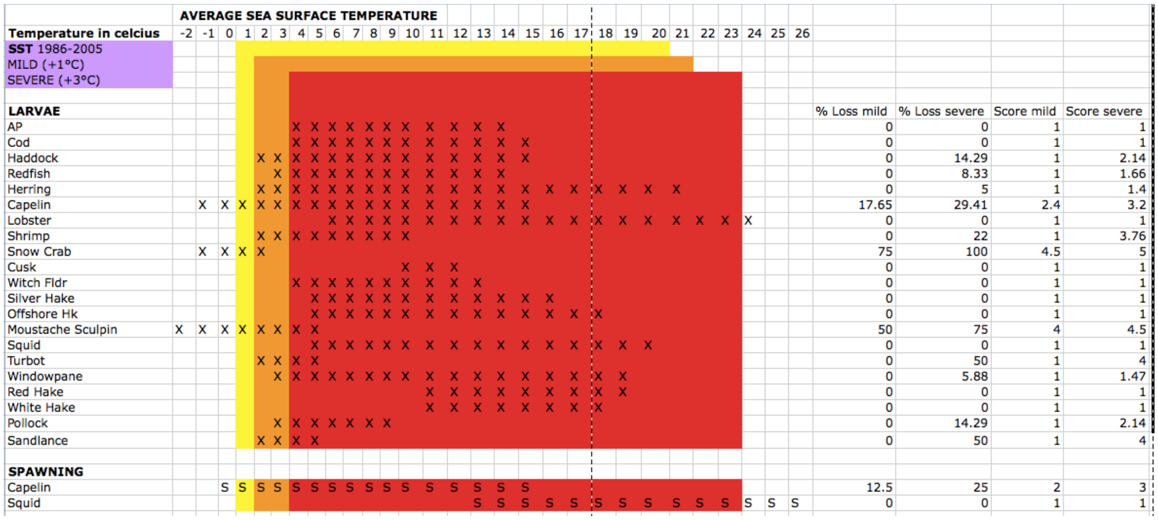


Figure B.2. These boxplots represent habitat change from SDM habitat model from normal (now) to severe warming scenario from 10 model fits using subsampled data for each Gadoid species. Regions are defined in Shackell *et al*., 2012: ESS = eastern Scotian Shelf, WSS = western Scotian Shelf, BoF = Bay of Fundy, GoM = Gulf of Maine, GB = Georges Bank, SNE = Southern New England, and MAB = Mid-Atlantic Bight. Various estimates for region/species combinations are missing indicating a poor model fit (Shackell *et al*., 2014).

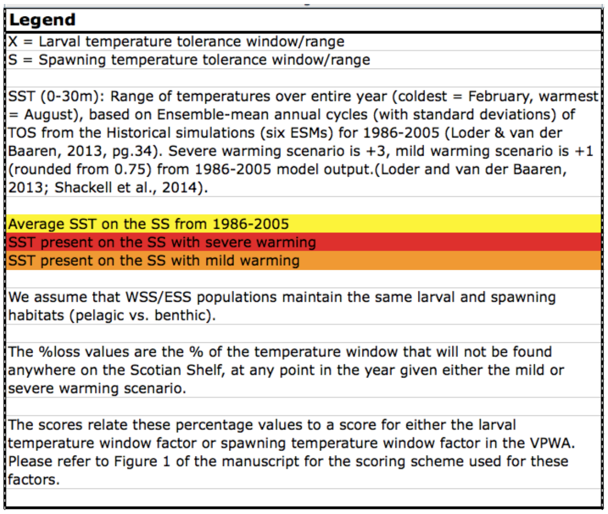
APPENDIX C: RAW DATA AND METHOD FOR LARVAL AND SPAWNING TEMPERATURE WINDOW VPWA FACTORS.

Simple analyses were used to derive sensitivity scores for larval and spawning temperature window factors (E2 and E3). Due to the fact that spatial data for larval, egg, and spawning life stages are limited, these analyses were used in lieu of the ability to use a species distribution model. Species temperature tolerance ranges were derived from the literature (refer to Appendix E and Appendix F). The aim was to obtain temperature ranges that corresponded to each species' entire geographical range to secure maximum upper and minimum lower thermal limits. The species shown in the tables below are the only species for which this information was available (Table C.1 and Table C.2). Other species were given a score of 3 for these factors in order to not over or under-estimate vulnerability (refer to Appendix E). The assumptions were that species thermal tolerance does not change among the ESS and WSS populations assessed, that where these life stages occur (i.e., pelagic vs. benthic) was consistent geographically. Further, due to a lack of abundance data to correspond to these thermal ranges, it was assumed that abundance of spawners and larvae are distributed evenly throughout the temperature window.

Table C.1. This table depicts the methods used to derive percent loss of thermal window for larval and spawning life stages, when those life stages occur at or near the ocean surface. Range of sea surface temperatures highlighted in yellow are the observed (1986-2005) averages from the coldest month of the year (February) to the warmest month of the year (August) on the Scotian Shelf (Loder and van der Baaren, 2013). Range of sea surface temperatures highlighted in orange are the observed +1°C (rounded from 0.7, derived from the mild warming scenario). Range of sea surface temperatures highlighted in red are the observed +3°C (severe warming scenario). The "x"s show the temperatures tolerated by species (from literature listed in Appendix E) at the larval stage. The "s"s show the temperatures tolerated by species (from literature listed in Appendix E) at the spawning and/or egg stage. The percent loss under the mild warming scenario is the percent of "x"s or "s"s not highlighted in orange (note that the upper bound of the orange highlighted columns is behind the red). The percent loss under the severe warming scenario is the percent of "x"s or "s"s not highlighted in red.



	AVERAGE SEA SURFACE TEMPERATURE																																
Temperature in celcius	-2	-1	0	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26				
SST 1986-2005				🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨										
MILD (+1°C)					🟧	🟧	🟧	🟧	🟧	🟧	🟧	🟧	🟧	🟧	🟧	🟧	🟧	🟧	🟧	🟧	🟧	🟧	🟧	🟧									
SEVERE (+3°C)							🟥	🟥	🟥	🟥	🟥	🟥	🟥	🟥	🟥	🟥	🟥	🟥	🟥	🟥	🟥	🟥	🟥	🟥	🟥	🟥							
LARVAE																														% Loss mild	% Loss severe	Score mild	Score severe
AP							X	X	X	X	X	X	X	X	X	X	X													0	0	1	1
Cod							X	X	X	X	X	X	X	X	X	X	X	X												0	0	1	1
Haddock					X	X	X	X	X	X	X	X	X	X	X	X	X	X												0	14.29	1	2.14
Redfish						X	X	X	X	X	X	X	X	X	X	X	X													0	8.33	1	1.66
Herring					X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X						0	5	1	1.4
Capelin		X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X												17.65	29.41	2.4	3.2
Lobster									X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X			0	0	1	1
Shrimp					X	X	X	X	X	X	X	X	X																	0	22	1	3.76
Snow Crab		X	X	X	X																									75	100	4.5	5
Cusk													X	X	X															0	0	1	1
Witch Fldr							X	X	X	X	X	X	X	X	X	X														0	0	1	1
Silver Hake								X	X	X	X	X	X	X	X	X	X	X	X											0	0	1	1
Offshore Hk								X	X	X	X	X	X	X	X	X	X	X	X	X	X									0	0	1	1
Moustache Sculpin	X	X	X	X	X	X	X	X																						50	75	4	4.5
Squid								X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X							0	0	1	1
Turbot					X	X	X	X																						0	50	1	4
Windowpane						X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X								0	5.88	1	1.47
Red Hake														X	X	X	X	X	X	X	X	X								0	0	1	1
White Hake														X	X	X	X	X	X	X	X									0	0	1	1
Pollock						X	X	X	X	X	X	X																		0	14.29	1	2.14
Sandlance					X	X	X	X																						0	50	1	4
SPAWNING																																	
Capelin			S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S												12.5	25	2	3
Squid																S	S	S	S	S	S	S	S	S	S	S	S	S	S	0	0	1	1



Legend

X = Larval temperature tolerance window/range
S = Spawning temperature tolerance window/range

SST (0-30m): Range of temperatures over entire year (coldest = February, warmest = August), based on Ensemble-mean annual cycles (with standard deviations) of TOS from the Historical simulations (six ESMs) for 1986-2005 (Loder & van der Baaren, 2013, pg.34). Severe warming scenario is +3, mild warming scenario is +1 (rounded from 0.75) from 1986-2005 model output.(Loder and van der Baaren, 2013; Shackell et al., 2014).

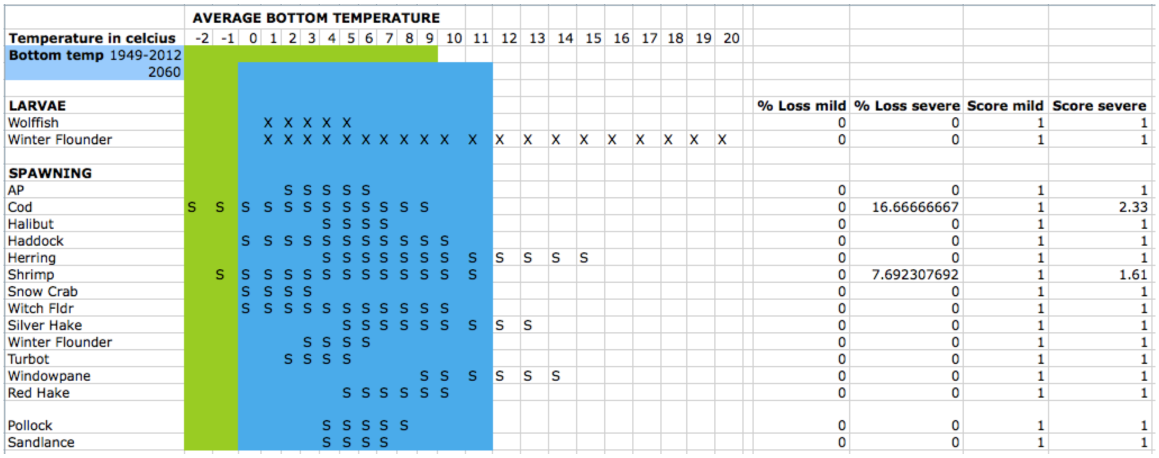
🟨 Average SST on the SS from 1986-2005
🟥 SST present on the SS with severe warming
🟧 SST present on the SS with mild warming

We assume that WSS/ESS populations maintain the same larval and spawning habitats (pelagic vs. benthic).

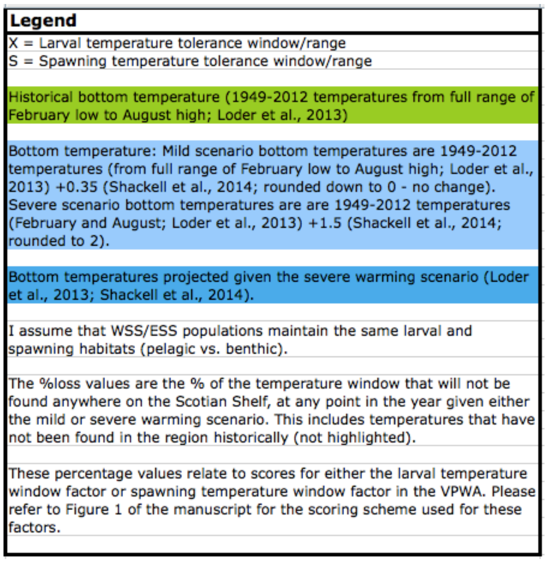
The %loss values are the % of the temperature window that will not be found anywhere on the Scotian Shelf, at any point in the year given either the mild or severe warming scenario.

The scores relate these percentage values to a score for either the larval temperature window factor or spawning temperature window factor in the VPWA. Please refer to Figure 1 of the manuscript for the scoring scheme used for these factors.

Table C.2. This table depicts the methods used to derive percent loss of thermal window for larval and spawning life stages, when those life stages occur on or near the ocean bottom. Range of bottom temperatures highlighted in green are the observed (1949-2005) averages from the coldest month of the year (February) and the warmest month of the year (August) on the Scotian Shelf (Loder *et al*., 2013). Projected warming for bottom habitat under the mild scenario is +0.35°C. This was rounded down to +0°C and therefore is not shown. Consequently, all species listed here are considered to not lose any of their thermal window under the mild warming scenario. Range of bottom temperatures highlighted in blue are the observed +3°C (severe warming scenario). The “x”s show the temperatures tolerated by species (from literature listed in Appendix E) at the larval stage. The “s”s show the temperatures tolerated by species (from literature listed in Appendix E) at the spawning and/or egg stage. The percent loss under the severe warming scenario is the percent of “x”s or “s”s not highlighted in blue.



	AVERAGE BOTTOM TEMPERATURE																										
Temperature in celcius	-2	-1	0	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20				
Bottom temp 1949-2012	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩															
2060			🟦	🟦	🟦	🟦	🟦	🟦	🟦	🟦	🟦	🟦	🟦	🟦													
LARVAE																								**% Loss mild**	**% Loss severe**	**Score mild**	**Score severe**
Wolffish				X	X	X	X	X																0	0	1	1
Winter Flounder				X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	0	0	1	1
SPAWNING																											
AP					S	S	S	S	S															0	0	1	1
Cod	S	S	S	S	S	S	S	S	S	S	S	S												0	16.66666667	1	2.33
Halibut							S	S	S	S														0	0	1	1
Haddock			S	S	S	S	S	S	S	S	S	S	S											0	0	1	1
Herring							S	S	S	S	S	S	S	S	S	S	S	S						0	0	1	1
Shrimp		S	S	S	S	S	S	S	S	S	S	S	S	S										0	7.692307692	1	1.61
Snow Crab			S	S	S	S																		0	0	1	1
Witch Fldr			S	S	S	S	S	S	S	S	S	S	S											0	0	1	1
Silver Hake								S	S	S	S	S	S	S	S	S								0	0	1	1
Winter Flounder						S	S	S	S															0	0	1	1
Turbot					S	S	S	S																0	0	1	1
Windowpane												S	S	S	S	S	S							0	0	1	1
Red Hake								S	S	S	S	S	S											0	0	1	1
Pollock							S	S	S	S	S													0	0	1	1
Sandlance							S	S	S	S														0	0	1	1



Legend

X = Larval temperature tolerance window/range
S = Spawning temperature tolerance window/range

🟩 Historical bottom temperature (1949-2012 temperatures from full range of February low to August high; Loder et al., 2013)

🟦 Bottom temperature: Mild scenario bottom temperatures are 1949-2012 temperatures (from full range of February low to August high; Loder et al., 2013) +0.35 (Shackell et al., 2014; rounded down to 0 - no change). Severe scenario bottom temperatures are are 1949-2012 temperatures (February and August; Loder et al., 2013) +1.5 (Shackell et al., 2014; rounded to 2).

🟦 Bottom temperatures projected given the severe warming scenario (Loder et al., 2013; Shackell et al., 2014).

I assume that WSS/ESS populations maintain the same larval and spawning habitats (pelagic vs. benthic).

The %loss values are the % of the temperature window that will not be found anywhere on the Scotian Shelf, at any point in the year given either the mild or severe warming scenario. This includes temperatures that have not been found in the region historically (not highlighted).

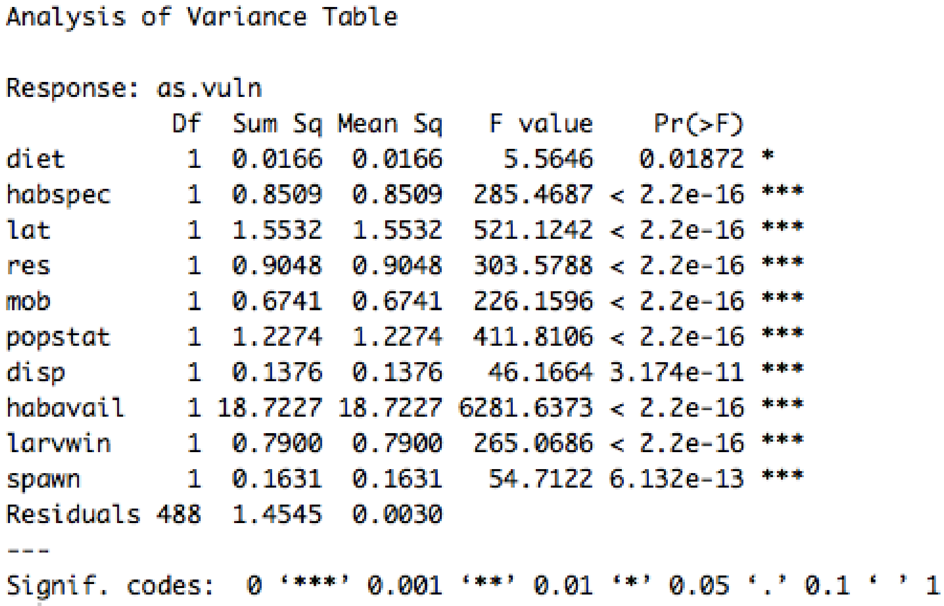
These percentage values relate to scores for either the larval temperature window factor or spawning temperature window factor in the VPWA. Please refer to Figure 1 of the manuscript for the scoring scheme used for these factors.

APPENDIX D: SENSITIVITY ANALYSIS

Table D.1. List of abbreviations for factor names

Factor name	Abbreviation
Projected habitat availability	habavail
Larval temperature window	larvwin
Spawning temperature window	spawn
Latitudinal range	lat
Habitat specificity	habspec
Diet specificity	diet
Population status	popstat
Resilience	res
Larval dispersal	disp
Adult mobility	mob

Table D.2. Results of an analysis of variance test (ANOVA), examining whether each factor within the CCVA has a significant effect on resultant vulnerability scores. Factor scores were randomly generated (between the minimum and maximum possible scores) for this analysis using a Monte Carlo simulation. The results indicate that all factors have a significant impact on the outcome of the model.



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Analysis of Variance Table

Response: as.vuln
           Df  Sum Sq Mean Sq   F value    Pr(>F)    
diet        1  0.0166  0.0166    5.5646   0.01872 *  
habspec     1  0.8509  0.8509  285.4687 < 2.2e-16 ***
lat         1  1.5532  1.5532  521.1242 < 2.2e-16 ***
res         1  0.9048  0.9048  303.5788 < 2.2e-16 ***
mob         1  0.6741  0.6741  226.1596 < 2.2e-16 ***
popstat     1  1.2274  1.2274  411.8106 < 2.2e-16 ***
disp        1  0.1376  0.1376   46.1664 3.174e-11 ***
habavail    1 18.7227 18.7227 6281.6373 < 2.2e-16 ***
larvwin     1  0.7900  0.7900  265.0686 < 2.2e-16 ***
spawn       1  0.1631  0.1631   54.7122 6.132e-13 ***
Residuals 488  1.4545  0.0030                      
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1
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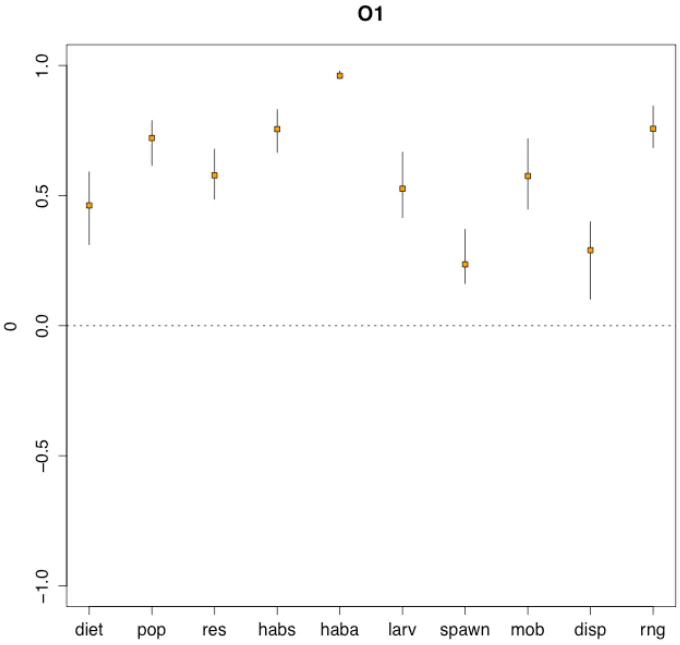

Figure D.1. Linear correlation coefficient for the effect of each factor on overall vulnerability. All factors have a positive, non-zero correlation with vulnerability.

APPENDIX E: RAW FACTOR SCORES AND REFERENCES FOR ALL SPECIES AND POPULATIONS ASSESSED

Table E.1. Raw factor scores for all 33 species (including ESS and WSS populations for seven of these species) with corresponding references. Factors are: adult projected thermal habitat availability (hab.avail), loss from larval thermal window (larval.window), loss from spawning thermal window, adult mobility (ad.mobility), larval dispersal (larv.disp), COSEWIC population status (pop.stat), life history resilience (resilience), diet specificity (diet.spec), habitat specificity (hab.spec), and latitudinal range (lat.range). The importance weighting of each factor is shown in the "weight" column. Species scores are given in the "score.mild" and "score.severe" columns for mild and severe warming scenarios respectively. Scores weighted by (multiplied or given the exponent of) factor importance weightings are given in columns "wscore.mild" and "wscore.severe". The references used to obtain these scores are given in the "source(s)" column. Full references are provided in Appendix F. The meaning of these factor scores can be reviewed in Figure 2.1 in the main text.

common name	component	factor	factor weight	score. mild	wscore .mild	score. severe	wscore. severe	source(s)
American Lobster	exposure	hab.avail	3	1	1	1	1	Shackell et al., 2014; OBIS-Canada
American Lobster	exposure	larval.window	2	1	1	1	1	Templeman, 1936; Caddy, 1979; Aiken & Waddy, 1980; MacKenzie, 1988
American Lobster	exposure	spawn.window	1	3	3	3	3	n/a
American Lobster	rigidity	ad.mobility	2	3	6	3	6	Cooper & Uzmann, 1971; Campbell, 1986; DFO, 1997; Tremblay et al., 2001
American Lobster	rigidity	larv.disp	1	2	2	2	2	Aiken & Waddy, 1980; Nichols & Thompson, 1988; DFO, 1997; Cobb & Wahle, 1994; Katz et al., 1994; Tremblay et al., 2001
American Lobster	rigidity	pop.stat	3	2	6	2	6	COSEWIC, 2012
American Lobster	rigidity	resilience	2	1	2	1	2	Aiken & Waddy, 1980; Campbell & Pezzack, 1986; Pezzack, 1992; Musick, 1999; Tremblay & Eagles, 1998; Agnalt, 2008
American Lobster	sensitivity	diet.spec	1	1	1	1	1	Carter & Steele, 1982; Elner & Campbell, 1987

American Lobster	sensitivity	hab.spec	2	3	6	3	6	Botero & Atema, 1982; Johns & Mann, 1987; Boudreau et al., 1990; Pezzack, 1992; Cobb & Wahle, 1994; Tremblay et al., 2001
American Lobster	sensitivity	lat.range	2	3.75	7.5	3.75	7.5	iobis.org
American Plaice	exposure	hab.avail	3	1.29	2.146689	2.4	13.824	Shackell et al., 2014; OBIS-Canada
American Plaice	exposure	larval.window	2	1	1	1	1	Howell & Caldwell, 1984; Johnson et al., 1999
American Plaice	exposure	spawn.window	1	1	1	1	1	n/a
American Plaice	rigidity	ad.mobility	2	4	8	4	8	Bowering & Brodie, 1991; Frimodt, 1995; Johnson et al., 1999; Coad & Reist, 2004; COSEWIC, 2009
American Plaice	rigidity	larv.disp	1	2	2	2	2	Russell, 1976; Johnson et al., 1999; Murua & Saborido-Rey, 2003
American Plaice	rigidity	pop.stat	3	5	15	5	15	COSEWIC, 2012
American Plaice	rigidity	resilience	2	3.6	7.2	3.6	7.2	Musick, 1999; COSEWIC, 2009
American Plaice	sensitivity	diet.spec	1	1	1	1	1	Pitt, 1973; Langton & Bowman, 1980; Berestovsky, 1989; Johnson et al., 1999; Bowman et al., 2000; Coad & Reist, 2004; ICES, 2012
American Plaice	sensitivity	hab.spec	2	2	4	2	4	Bowering & Brodie, 1991; Johnson et al., 1999; Coad & Reist, 2004; COSEWIC, 2009; Morgan et al., 2011
American	sensitivity	lat.range	2	2.3	4.6	2.3	4.6	Froese & Pauly,

Plaice									2000; OBIS-Canada
Atlantic Herring	exposure	hab.avail	3	1.3	2.197	3.1	29.791	Shackell et al., 2014; OBIS-Canada
Atlantic Herring	exposure	larval.window	2	1	1	1.4	1.96	Reid et al., 1999; Oeberst et al., 2009
Atlantic Herring	exposure	spawn.window	1	1	1	1	1	Reid et al., 1999; Portner & Peck, 2010
Atlantic Herring	rigidity	ad.mobility	2	2	4	2	4	Blaxter, 1990; Reid et al., 1999; Corten, 2002; Melvin & Power, 2008; DFO, 2011a
Atlantic Herring	rigidity	larv.disp	1	2	2	2	2	Muus & Dahlstrom, 1974; Russell, 1976; Blaxter, 1990; Thiel et al., 1996; Reid, 1999; Melvin & Power, 2008
Atlantic Herring	rigidity	pop.stat	3	1	3	1	3	COSEWIC, 2012
Atlantic Herring	rigidity	resilience	2	2.3	4.6	2.3	4.6	COSEWIC, Whitehead, 1985; Blaxter, 1990; Reid et al., 1999; Musick, 1999
Atlantic Herring	sensitivity	diet.spec	1	4	4	4	4	Rice, 1963; Last, 1987; Blaxter, 1990; Thiel et al., 1996; Reid et al., 1999; Bowman et al., 2000; Melvin & Power, 2008
Atlantic Herring	sensitivity	hab.spec	2	1	2	1	2	Whitehead, 1984; Whitehead, 1985; Blaxter, 1990; Reid et al., 1999; Melvin & Power, 2008
Atlantic Herring	sensitivity	lat.range	2	3.45	6.9	3.45	6.9	Froese & Pauly, 2000; OBIS-Canada
Atlantic Wolffish	exposure	hab.avail	3	1.04	1.124864	3.14	30.959144	Shackell et al., 2014; OBIS-Canada
Atlantic Wolffish	exposure	larval.window	2	1	1	1	1	Pavlov & Novikov, 1993; Cargnelli et al., 1999; Shackel & Frank, 2000

Atlantic Wolffish	exposure	spawn.window	1	3	3	3	3	n/a
Atlantic Wolffish	rigidity	ad.mobility	2	2	4	2	4	Bigelow & Schroeder, 1953; Barsukov, 1986; Johannessen et al., 1993; Coad & Reist, 2004; Kulka et al., 2007 (review of more than 10 relevant references); McCusker & Bentzen, 2010
Atlantic Wolffish	rigidity	larv.disp	1	4	4	4	4	Bigelow & Schroeder, 1953; Barsukov, 1986; Scott & Scott, 1988; Moksness & Pavlov 1996; Cargnelli et al., 1999; Kulka et al., 2007
Atlantic Wolffish	rigidity	pop.stat	3	3	9	3	9	COSEWIC, 2012
Atlantic Wolffish	rigidity	resilience	2	3.6	7.2	3.6	7.2	Keats et al., 1985; Barsukov, 1986; Johannessen et al., 1993; Pavlov & Novikov, 1993; Frimodt, 1995; Musick, 1999; COSEWIC, 2000; Coad & Reist, 2004; Kulka et al., 2007; Gunnarsson et al., 2008; Larocque et al., 2008; Simon et al., 2012
Atlantic Wolffish	sensitivity	diet.spec	1	1	1	1	1	Ortova et al., 1990; Bowman et al., 2000; Monk, 2002; Kulka et al., 2007 (review of more than 10 relevant references); ICES, 2012
Atlantic Wolffish	sensitivity	hab.spec	2	3	6	3	6	Barsukov, 1986; Johannessen et al., 1993; Frimodt, 1995; COSEWIC, 2000; Coad & Reist, 2004; Kulka

									et al., 2007 (review of more than 10 relevant references); Larocque et al., 2008; Simon et al., 2012
Atlantic Wolffish	sensitivity	lat.range	2	2.75	5.5	2.75	5.5	OBIS-Canada (iobis.org; survey and observer data from various sources)
Blackbelly Rosefish	exposure	hab.avail	3	1	1	1	1	Shackell et al., 2014; OBIS-Canada
Blackbelly Rosefish	exposure	larval.window	2	3	9	3	9	n/a
Blackbelly Rosefish	exposure	spawn.window	1	3	3	3	3	n/a
Blackbelly Rosefish	rigidity	ad.mobility	2	3	6	3	6	Eschmeyer & Dempster, 1990; Sedberry et al., 2006
Blackbelly Rosefish	rigidity	larv.disp	1	2	2	2	2	Wourms, 1991; White et al., 1998; Munoz et al., 1999; Sequeira et al., 2003; Sedberry et al., 2006
Blackbelly Rosefish	rigidity	pop.stat	3	1	3	1	3	COSEWIC, 2012
Blackbelly Rosefish	rigidity	resilience	2	5	10	5	10	Wourms, 1991; White et al., 1998; Kelly et al., 1999; Munoz et al., 1999; Musick, 1999; Allain & Lorance, 2000; Sequeira et al., 2012
Blackbelly Rosefish	sensitivity	diet.spec	1	1	1	1	1	Maurer & Bowman, 1975; Hureau & Litvinenko, 1986; Bowman et al., 2000; Smith & Link, 2010
Blackbelly Rosefish	sensitivity	hab.spec	2	3	6	3	6	Hureau et al., 1986; Eschmeyer & Dempster, 1990; Kelly et al., 1999; Sedberry et al., 2006
Blackbelly	sensitivity	lat.range	2	3.3	6.6	3.3	6.6	Froese & Pauly,

Rosefish								2000; OBIS-Canada
Capelin	exposure	hab.avail	3	2.7	19.683	4.22	75.151448	Shackell et al., 2014; OBIS-Canada
Capelin	exposure	larval.window	2	2.4	5.76	3.2	10.24	Leggett et al. 1984; Elliott & Leggett, 1995; Shackell & Frank, 2000; Doyle et al., 2002
Capelin	exposure	spawn.window	1	2	2	3	3	Bigelow & Schroeder, 1953; Frank & Leggett, 1981; Scott & Scott, 1988; Nakashima & Wheeler, 2002; Rose, 2005; Penton et al., 2012
Capelin	rigidity	ad.mobility	2	2	4	2	4	Hurtubise, 1993; Shackell et al., 1994; Pietsch et al., 2000; Rose, 2005; Huse & Ellingsen, 2008
Capelin	rigidity	larv.disp	1	2	2	2	2	Hurtubise, 1993; Shackell et al., 1994; Pietsch et al., 2000; Nakashima, 2002; Murua & Saborido-Rey, 2003; Rose, 2005; Penton et al., 2013
Capelin	rigidity	pop.stat	3	1	3	1	3	COSEWIC, 2012
Capelin	rigidity	resilience	2	2.3	4.6	2.3	4.6	Breder & Rosen, 1966; Beverton et al., 1985; Hurtubise, 1993; Hedeholm et al., 2011
Capelin	sensitivity	diet.spec	1	2	2	2	2	Kuznetsova, 1997; Orlova et al., 2005
Capelin	sensitivity	hab.spec	2	4	8	4	8	Hurtubise, 1993; Shackell et al., 1994; Pietsch et al., 2000; Nakashima, 2002; Rose, 2005; Penton et al., 2012
Capelin	sensitivity	lat.range	2	3.25	6.5	3.25	6.5	OBIS-Canada
ESS Cod	exposure	hab.avail	3	1	1	2.48	15.252992	Shackell pers comm.
ESS Cod	exposure	larval.window	2	1	1	1	1	Jordaan & Kling,

								2003; Bogstad et al., 2013
ESS Cod	exposure	spawn.window	1	1	1	2.33	2.33	Galloway et al., 1998; Johansen and Krogh 1914; Valerio et al. 1992; Brander & Hurley, 1992
ESS Cod	rigidity	ad.mobility	2	2	4	2	4	Fahay et al., 1999; Jordaan & Kling, 2003; Drinkwater, 2005
ESS Cod	rigidity	larv.disp	1	2	2	2	2	Muus & Dahlstrom, 1974; Russell, 1976; Robb, 1981; Fahay et al., 1999
ESS Cod	rigidity	pop.stat	3	5	15	5	15	COSEWIC, 2012
ESS Cod	rigidity	resilience	2	2.3	4.6	2.3	4.6	Cohen et al., 1990; Fahay et al., 1999; Musick, 1999; Dutil & Brander, 2003
ESS Cod	sensitivity	diet.spec	1	1	1	1	1	Artz, 1978; Langton & Bowman, 1980; Hacunda, 1981; Armstrong, 1982; Berestovsky, 1989; dos Santos & Falk-Petersen, 1989; Nordeide & Fossa, 1992; Greenstreet, 1996; Bromley et al., 1997; Bowman et al., 2000; Herrmann, 2004; ICES, 2012
ESS Cod	sensitivity	hab.spec	2	1	2	1	2	Cohen et al., 1990; Fahay et al., 1999; Drinkwater, 2005
ESS Cod	sensitivity	lat.range	2	2.2	4.4	2.2	4.4	iobis.org
ESS Cusk	exposure	hab.avail	3	1	1	1	1	Shackell et al., 2014; OBIS-Canada
ESS Cusk	exposure	larval.window	2	1	1	1	1	Shackell & Frank, 2000
ESS Cusk	exposure	spawn.window	1	3	3	3	3	n/a
ESS Cusk	rigidity	ad.mobility	2	4	8	4	8	Cohen et al., 1990; Bergstad, 1991; Riede, 2004
ESS Cusk	rigidity	larv.disp	1	2	2	2	2	Russell, 1976; Cohen et al., 1990

ESS Cusk	rigidity	pop.stat	3	5	15	5	15	COSEWIC, 2012
ESS Cusk	rigidity	resilience	2	3.6	7.2	3.6	7.2	Cohen et al., 1990; Musick, 1999
ESS Cusk	sensitivity	diet.spec	1	1	1	1	1	Langton & Bowman, 1980; Cohen et al., 1990; Bergstad, 1991; Bowman et al., 2000; ICES, 2012
ESS Cusk	sensitivity	hab.spec	2	2	4	2	4	Svetovidov, 1986; Cohen et al., 1990; Bergstad, 1991
ESS Cusk	sensitivity	lat.range	2	4.45	8.9	4.45	8.9	iobis.org
ESS Haddock	exposure	hab.avail	3	1	1	1	1	Shackell et al., 2014; OBIS-Canada
ESS Haddock	exposure	larval.window	2	1	1	2.14	4.5796	Laurence, 1978; Cargnelli et al., 1999; Martell et al., 2005; Neuheimer & Taggart, 2010; Bogstad et al., 2013
ESS Haddock	exposure	spawn.window	1	1	1	1	1	Page and Frank, 1989; Neuheimer & Taggart, 2010
ESS Haddock	rigidity	ad.mobility	2	2	4	2	4	Cohen et al., 1990; Cargnelli et al., 1999; DFO, 2002
ESS Haddock	rigidity	larv.disp	1	2	2	2	2	Russell, 1976; Hurley & Campana, 1989; Brander & Hurley, 1992; Cargnelli et al., 1999
ESS Haddock	rigidity	pop.stat	3	1	3	1	3	COSEWIC, 2012
ESS Haddock	rigidity	resilience	2	2.3	4.6	2.3	4.6	Muus & Dahlstrom, 1978; Cargnelli et al., 1999; Musick, 1999; Froese & Pauly, 2000; Neuheimer & Taggart, 2010
ESS Haddock	sensitivity	diet.spec	1	2	2	2	2	Marak, 1960; Muus & Dahlstrom, 1978; Langton & Bowman, 1980; Robb & Hislop, 1980; Robb, 1981; Daan, 1989; Greenstreet, 1996;

								Jiang & Jorgensen, 1996; Bromley et al., 1997; Cargnelli et al., 1999; ICES 2012
ESS Haddock	sensitivity	hab.spec	2	1	2	1	2	Scott, 1982; Scott & Scott, 1988; Hurley & Campana, 1989; Brander & Hurley, 1992: Cargnelli et al., 1999; DFO, 2002
ESS Haddock	sensitivity	lat.range	2	3.75	7.5	3.75	7.5	iobis.org
ESS Offshore Hake	exposure	hab.avail	3	1	1	4	64	Shackell et al., 2014; OBIS-Canada
ESS Offshore Hake	exposure	larval.window	2	1	1	1	1	Chang et al., 1999 (Reid et al., 1999)
ESS Offshore Hake	exposure	spawn.window	1	3	3	3	3	n/a
ESS Offshore Hake	rigidity	ad.mobility	2	3	6	3	6	Cohen et al., 1990; Chang et al., 1999 (Bigelow & Schroeder, 1955); Lloris et al., 2005
ESS Offshore Hake	rigidity	larv.disp	1	1	1	1	1	Chang et al., 1999 (Morse, 1989; Reid et al., 1999); Lloris et al., 2005
ESS Offshore Hake	rigidity	pop.stat	3	3	9	3	9	COSEWIC, 2012
ESS Offshore Hake	rigidity	resilience	2	2.3	4.6	2.3	4.6	Cohen et al., 1990; Lloris et al., 2005
ESS Offshore Hake	sensitivity	diet.spec	1	2	2	2	2	Langton & Bowman, 1980; Chang et al., 1999 (Bigelow & Schroeder, 1955; Langton & Bowman, 1980; Cohen et al., 1990); Bowman et al., 2000
ESS Offshore Hake	sensitivity	hab.spec	2	1	2	1	2	Cohen et al., 1990; Chang et al., 1999 (); Lloris et al., 2005
ESS	sensitivity	lat.range	2	1	2	1	2	iobis.org

Offshore Hake								
ESS Pollock	exposure	hab.avail	3	1.62	4.251528	4.58	96.071912	Shackell et al., 2014; OBIS-Canada
ESS Pollock	exposure	larval.window	2	1	1	2.14	4.5796	Cargnelli et al., 1999 (Bigelow & Schroeder, 1953; Hardy, 1978)
ESS Pollock	exposure	spawn.window	1	1	1	1	1	Collette & Klein-MacPhee, in prep.
ESS Pollock	rigidity	ad.mobility	2	2	4	2	4	Cohen et al., 1990; Frimodt, 1995; Bromley et al., 1997; Cargnelli et al., 1999 (Collette & Klein-MacPhee, in prep.; Hardy, 1978; Ojeda and Dearborn 1990; Rangeley and Kramer 1995)
ESS Pollock	rigidity	larv.disp	1	1	1	1	1	Cargnelli et al., 1999 (Bigelow & Schroeder, 1953; Hardy, 1978)
ESS Pollock	rigidity	pop.stat	3	2	6	2	6	COSEWIC, 2012
ESS Pollock	rigidity	resilience	2	2.3	4.6	2.3	4.6	Cargnelli et al., 1999 (Steele, 1963; Mayo, 1994); Musick, 1999; Froese & Pauly, 2000
ESS Pollock	sensitivity	diet.spec	1	4	4	4	4	Marak, 1960; Robb, 1981; Greenstreet, 1996; Bromley et al., 1997; Cargnelli et al., 1999; Bowman et al., 2000
ESS Pollock	sensitivity	hab.spec	2	2	4	2	4	Cohen et al., 1990; Frimodt, 1995; Cargnelli et al., 1999 (Hardy, 1978; Scott, 1982)
ESS Pollock	sensitivity	lat.range	2	3.8	7.6	3.8	7.6	iobis.org
ESS Red Hake	exposure	hab.avail	3	1	1	2.3	12.167	Shackell et al., 2014; OBIS-Canada
ESS Red Hake	exposure	larval.window	2	1	1	1	1	Steimle et al., 2000
ESS Red	exposure	spawn.window	1	1	1	1	1	Steimle et al.,

Hake									1999 (Wilk et al., 1990)
ESS Red Hake	rigidity	ad.mobility	2	2	4	2	4	Cohen et al., 1990; Frimodt, 1995; Steimle et al., 1999 (Bigelow & Schroeder, 1953; Tyler, 1971; Murawski & Finn, 1988; Jury et al., 1994; Stone et al., 1994; Able & Fahay, 1998)
ESS Red Hake	rigidity	larv.disp	1	1	1	1	1	Scotton et al., 1973; Svetovidov, 1986; Steimle et al., 1999
ESS Red Hake	rigidity	pop.stat	3	2	6	2	6	COSEWIC, 2012
ESS Red Hake	rigidity	resilience	2	2.3	4.6	2.3	4.6	Scotton et al., 1973; Svetovidov, 1986; Frimodt, 1995;Musick, 1999; Steimle et al., 1999; Froese & Pauly, 2000
ESS Red Hake	sensitivity	diet.spec	1	1	1	1	1	Langton & Bowman, 1980; Hacunda, 1981; Steimle et al., 1999; Bowman et al., 2000
ESS Red Hake	sensitivity	hab.spec	2	3	6	3	6	Cohen et al., 1990; Steimle et al., 1999 (Collette & Klein-MacPhee, in prep.; Gottschall et al., in review; Reid et al., 1979)
ESS Red Hake	sensitivity	lat.range	2	4.1	8.2	4.1	8.2	iobis.org
ESS Silver Hake	exposure	hab.avail	3	1	1	1	1	Shackell et al., 2014; OBIS-Canada
ESS Silver Hake	exposure	larval.window	2	1	1	1	1	Morse et al., 1999
ESS Silver Hake	exposure	spawn.window	1	1	1	1	1	Morse et al., 1999 (Schroeder, 1953; Fahay, 1974)
ESS Silver Hake	rigidity	ad.mobility	2	2	4	2	4	Scott & Scott, 1988; Cohen et al., 1990; Hesler et al., 1995; Lloris et al., 2005
ESS Silver	rigidity	larv.disp	1	1	1	1	1	Hesler et al., 1995;

Hake								Morse et al., 1999; Lloris et al., 2005
ESS Silver Hake	rigidity	pop.stat	3	2	6	2	6	COSEWIC, 2012
ESS Silver Hake	rigidity	resilience	2	2.3	4.6	2.3	4.6	Vinogradov & Kozachenko, 1989; Cohen et al., 1990; Hesler et al., 1995; Morse et al., 1999; Musick, 1999; Froese & pauly, 2000; Lloris et al., 2005
ESS Silver Hake	sensitivity	diet.spec	1	2	2	2	2	Langton & Bowman, 1980; Bowman, 1984; Vinogradov & Kozachenko, 1989; Morse et al., 1999; Bowman et al., 2000
ESS Silver Hake	sensitivity	hab.spec	2	2	4	2	4	Cohen et al., 1990; Hesler et al., 1995; Morse et al., 1999; Lloris et al., 2005
ESS Silver Hake	sensitivity	lat.range	2	3.6	7.2	3.6	7.2	iobis.org
Halibut	exposure	hab.avail	3	1	1	1	1	Shackell et al., 2014; OBIS-Canada
Halibut	exposure	larval.window	2	3	9	3	9	n/a
Halibut	exposure	spawn.window	1	1	1	1	1	Bowering, 1986; Miller et al., 1991
Halibut	rigidity	ad.mobility	2	1	2	1	2	Nielsen, 1986; Cargnelli et al., 1999; Kohl Kanwit, 2007; DFO, 2011
Halibut	rigidity	larv.disp	1	2	2	2	2	Russell, 1976; Haug, 1990; Breder & Rosen, 1996; Cargnelli et al., 1999
Halibut	rigidity	pop.stat	3	3	9	3	9	COSEWIC, 2012
Halibut	rigidity	resilience	2	3.6	7.2	3.6	7.2	Muus & Dahlstrom, 1974; Nielsen, 1986; Haug, 1990; Cargnelli et al., 1999; Musick, 1999; Froese & Pauly, 2000
Halibut	sensitivity	diet.spec	1	1	1	1	1	Muus & Dahlstrom, 1974; Nielsen, 1986;

								Haug, 1990; Cargnelli et al., 1999; Bowman et al., 2000; Bundy, 2005
Halibut	sensitivity	hab.spec	2	3	6	3	6	Nielsen, 1986; Cargnelli et al., 1999; Muus & Nielsen, 1999;
Halibut	sensitivity	lat.range	2	2.9	5.8	2.9	5.8	iobis.org
Little Skate	exposure	hab.avail	3	1.46	3.112136	3.05	28.372625	Shackell et al., 2014; OBIS-Canada
Little Skate	exposure	larval.window	2	3	9	3	9	n/a
Little Skate	exposure	spawn.window	1	3	3	3	3	n/a
Little Skate	rigidity	ad.mobility	2	3	6	3	6	Frisk, 2002; Carrier et al., 2010
Little Skate	rigidity	larv.disp	1	4	4	4	4	McEachran & Dunn, 1998; Bor, 2002; McPhie & Campana, 2009; Carrier et al., 2010
Little Skate	rigidity	pop.stat	3	2	6	2	6	COSEWIC, 2012
Little Skate	rigidity	resilience	2	3.6	7.2	3.6	7.2	Breder & Rosen, 1966; McEachran & Dunn, 1998; Musick, 1999; Froese & Pauly, 2000; Frisk, 2002; McPhie & Campana, 2009; Carrier et al., 2010
Little Skate	sensitivity	diet.spec	1	4	4	4	4	Hacunda, 1981; McEachran & Dunn, 1998; Bowman et al., 2000
Little Skate	sensitivity	hab.spec	2	3	6	3	6	Robins & Ray, 1986; Bratton & Ayres, 1987; Scott & Scott, 1988; Carrier et al., 2010
Little Skate	sensitivity	lat.range	2	4.1	8.2	4.1	8.2	iobis.org
Longhorn Sculpin	exposure	hab.avail	3	1.35	2.460375	2.88	23.887872	Shackell et al., 2014; OBIS-Canada
Longhorn Sculpin	exposure	larval.window	2	3	9	3	9	n/a
Longhorn Sculpin	exposure	spawn.window	1	3	3	3	3	n/a
Longhorn Sculpin	rigidity	ad.mobility	2	4	8	4	8	Fahay, 1983; Robins & Ray, 1986; Scott & Scott, 1988

Longhorn Sculpin	rigidity	larv.disp	1	1	1	1	1	Warfel & Merriman, 1944; Altman & Dittmer, 1962; Fahay, 1983; Robins & Ray, 1986
Longhorn Sculpin	rigidity	pop.stat	3	2	6	2	6	COSEWIC, 2012
Longhorn Sculpin	rigidity	resilience	2	1	2	1	2	Warfel & Merriman, 1944; Altman & Dittmer, 1962; Robins & Ray, 1986; Musick, 1999; Froese & Pauly, 2000
Longhorn Sculpin	sensitivity	diet.spec	1	1	1	1	1	Hacunda, 1981; Laroche, 1982; Bowman et al., 2000; Link & Almeida, 2002
Longhorn Sculpin	sensitivity	hab.spec	2	3	6	3	6	Warfel & Merriman, 1944; Altman & Dittmer, 1962; Fahay, 1983; Munehara, 1992
Longhorn Sculpin	sensitivity	lat.range	2	3.75	7.5	3.75	7.5	Froese & Pauly, 2000; OBIS-Canada
Monkfish	exposure	hab.avail	3	1	1	1	1	Shackell et al., 2014; OBIS-Canada
Monkfish	exposure	larval.window	2	3	9	3	9	n/a
Monkfish	exposure	spawn.window	1	3	3	3	3	n/a
Monkfish	rigidity	ad.mobility	2	3	6	3	6	Robins & Ray, 1986; Laurenson et al., 2005; Johnson et al., 2007
Monkfish	rigidity	larv.disp	1	2	2	2	2	Fahay, 1983; Armstrong et al., 1992; Johnson et al., 2007
Monkfish	rigidity	pop.stat	3	2	6	2	6	COSEWIC, 2012
Monkfish	rigidity	resilience	2	3.6	7.2	3.6	7.2	Beverton & Holt, 1959; Robins & Ray, 1986; Armstrong et al., 1992; Musick, 1999; Froese & Pauly, 2000; Johnson et al., 2007

Monkfish	sensitivity	diet.spec	1	2	2	2	2	Scott & Scott, 1988; Bowman et al., 2000
Monkfish	sensitivity	hab.spec	2	1	2	1	2	Scott & Scott, 1988; Robins & Ray, 1986; Johnson et al., 2007
Monkfish	sensitivity	lat.range	2	3.2	6.4	3.2	6.4	iobis.org
Moustache Sculpin	exposure	hab.avail	3	3.6	46.656	4.85	114.084125	Shackell et al., 2014; OBIS-Canada
Moustache Sculpin	exposure	larval.window	2	4	16	4.5	20.25	Shackell & Frank, 2000
Moustache Sculpin	exposure	spawn.window	1	3	3	3	3	n/a
Moustache Sculpin	rigidity	ad.mobility	2	4	8	4	8	Pietsch, 1993; Similar species, Longhorn Sculpin (Fahay, 1983; Robins & Ray, 1986; Scott & Scott, 1988)
Moustache Sculpin	rigidity	larv.disp	1	2	2	2	2	Laroche, 1982; Fahay, 1983
Moustache Sculpin	rigidity	pop.stat	3	1	3	1	3	COSEWIC, 2012
Moustache Sculpin	rigidity	resilience	2	3.6	7.2	3.6	7.2	Pietsch, 1993; Musick, 1999; Froese & Pauly, 2000
Moustache Sculpin	sensitivity	diet.spec	1	2	2	2	2	Laroche, 1982; Muus & Nielsen, 1999; Bowman et al., 2000
Moustache Sculpin	sensitivity	hab.spec	2	3	6	3	6	Leim & Scott, 1966; Pietsch, 1993; Muus & Nielsen, 1999; Similar species, Longhorn Sculpin (Warfel & Merriman, 1944; Altman & Dittmer, 1962; Fahay, 1983; Munehara, 1992)
Moustache Sculpin	sensitivity	lat.range	2	3.05	6.1	3.05	6.1	iobis.org
Northern Shortfin Squid	exposure	hab.avail	3	1	1	1	1	Shackell et al., 2014; OBIS-Canada
Northern Shortfin Squid	exposure	larval.window	2	1	1	1	1	Hendrickson & Holmes, 2005 (Vecchione, 1979;

								O'Dor, 1983; Dawe & Beck, 1985; Hatanaka et al., 1985; Vicchione & Roper, 1986)
Northern Shortfin Squid	exposure	spawn.window	1	1	1	1	1	Hendrickson & Holmes, 2005 (O'Dor et al., 1982; Balch et al., 1985)
Northern Shortfin Squid	rigidity	ad.mobility	2	1	2	1	2	Hendrickson & Holmes, 2005 (Dawe et al., 1981; Dawe & Beck, 1985; Hatanaka et al., 1985; Rowell et al., 1985; Black et al., 1987; Hendricksonm, 2004)
Northern Shortfin Squid	rigidity	larv.disp	1	3	3	3	3	Hendrickson & Holmes, 2005 (Roper & Lu, 1979; Trites, 1983; Dawe & Beck, 1985; Hatanaka et al., 1985; Rowell et al., 1985)
Northern Shortfin Squid	rigidity	pop.stat	3	2	6	2	6	COSEWIC, 2012
Northern Shortfin Squid	rigidity	resilience	2	1	2	1	2	Hendrickson & Holmes, 2005 (Black et al., 1987; Caddy, 1991; Jackson, 1994; Perez, 1994; O'Dor & Dawe, 1998)
Northern Shortfin Squid	sensitivity	diet.spec	1	1	1	1	1	Hendrickson & Holmes, 2005 (Squires, 1957; O'Dor et al., 1980; Froerman, 1984; Vinogradov, 1984; Maurer & Bowman, 1985; Dawe, 1988; Dawe et al., 1997; Dawe & Brodziak, 1998)
Northern	sensitivity	hab.spec	2	2	4	2	4	Hendrickson &

Shortfin Squid									Holmes, 2005 (Amaratunga et al. 1980a; Fedulov and Froerman 1980; Dawe and Beck 1985; Hatanaka et al. 1985; Rowell et al. 1985; Perez 1994; Felley and Vecchione 1995; O'Dor and Dawe 1998; Hendrickson 2004)
Northern Shortfin Squid	sensitivity	lat.range	2	3.25	6.5	3.25	6.5	iobis.org
Northern Shrimp	exposure	hab.avail	3	1.32	2.299968	2.61	17.779581	Shackell et al., 2014; OBIS-Canada
Northern Shrimp	exposure	larval.window	2	1	1	3.76	14.1376	Shumway et al., 1985; Chabot & Ouellet, 2005; Ouellet & Chabot, 2005; Arnberg et al., 2012
Northern Shrimp	exposure	spawn.window	1	1	1	1.61	1.61	Allen, 1959; Shumway et al. 1985; Bergstrom 2000; Garcia, 2007
Northern Shrimp	rigidity	ad.mobility	2	3	6	3	6	Shumway et al., 1985 (Wollebaek, 1903; Berkeley, 1930; Hjort & Ruud, 1938; Horsted & Smidt, 1956; Apollonio & Dunton, 1969; Haynes & Wigley, 1969; Barr, 1970; Dow, 1981; Spencer et al., 1986); DFO, 2012
Northern Shrimp	rigidity	larv.disp	1	1	1	1	1	Shumway et al., 1985 (Berkeley, 1930; Hjort & Ruud, 1938; Horsted & Smidt, 1956; Apollonio & Dunton, 1969; Haynes & Wigley, 1969); Pedersen et al., 2003; DFO,

								2012
Northern Shrimp	rigidity	pop.stat	3	2	6	2	6	COSEWIC, 2012
Northern Shrimp	rigidity	resilience	2	2.3	4.6	2.3	4.6	Shumway et al., 1985 (Rasmussen, 1953; Horsted & Smidt, 1956; Allen, 1959; Apollonio & Dunton, 1969; Haynes & Wigley, 1969; Ito, 1976); DFO, 2012
Northern Shrimp	sensitivity	diet.spec	1	1	1	1	1	Shumway et al., 1985 (Wollebaek, 1903; Mistakidis, 1957; Butler, 1964; Barr, 1970; Dahlstrom, 1970; Fox, 1972; Feder et al., 1978)
Northern Shrimp	sensitivity	hab.spec	2	3	6	3	6	Shumway et al., 1985 (Wollebaek 1908; Hjort and Ruud 1938; Bigelow and Schroeder 1939; Horsted and Smidt 1956; Ivanov 1967; Rasmussen 1953; Smidt 1967; Warren and Sheldon 1968; Haynes and Wigley 1969; Barr 1970); DFO, 2012
Northern Shrimp	sensitivity	lat.range	2	4.25	8.5	4.25	8.5	iobis.org
Ocean Pout	exposure	hab.avail	3	2	8	4.15	71.473375	Shackell et al., 2014; OBIS-Canada
Ocean Pout	exposure	larval.window	2	3	9	3	9	n/a
Ocean Pout	exposure	spawn.window	1	3	3	3	3	n/a
Ocean Pout	rigidity	ad.mobility	2	2	4	2	4	Steimle et al., 1999 (Bigelow and Schroeder 1953; Sheehy et al. 1977; Wigley 1998)
Ocean Pout	rigidity	larv.disp	1	4	4	4	4	Steimle et al., 1999 (Jury et al., 1994)
Ocean Pout	rigidity	pop.stat	3	2	6	2	6	COSEWIC, 2012
Ocean Pout	rigidity	resilience	2	3.6	7.2	3.6	7.2	Musick, 1999; Steimle et al.,

									1999 (Olsen & merriman, 1946; O'Brien et al., 1993); Froese & Pauly, 2000
Ocean Pout	sensitivity	diet.spec	1	1	1	1	1	Steimle et al., 1999 (Clemens & Clemens, 1921; Olsen & Merriman, 1946; Orach-Meza, 1975; Hacunda, 1981; Clark & Livingstone, 1982; Buzulutskaya, 1983; MacDonald, 1983; Sedberry, 1983; MacDonald & Green, 1986)
Ocean Pout	sensitivity	hab.spec	2	3	6	3	6	Steimle et al., 1999 (Smith 1898; Olsen and Merriman 1946; Bigelow and Schroeder 1953; Orach-Meza 1975; Sheehy et al., 1977; Clark and Livingstone 1982; Keats et al. 1985; Able et al. 1987; Eklund 1988; Scott and Scott 1988; Jury et al. 1994; Auster et al., 1995;
Ocean Pout	sensitivity	lat.range	2	3.75	7.5	3.75	7.5	iobis.org
Redfish spp.	exposure	hab.avail	3	1	1	1.15	1.520875	Shackell et al., 2014; OBIS-Canada
Redfish spp.	exposure	larval.window	2	1	1	1.66	2.7556	Pikanowski et al., 1999; Sevigny et al., 2008; COSEWIC, 2010
Redfish spp.	exposure	spawn.window	1	3	3	3	3	n/a
Redfish spp.	rigidity	ad.mobility	2	4	8	4	8	Pikanowski et al., 1999 (Kelly & Barker, 1961; Kenchington, 1984; Murawski, 1993); Sevigny et al., 2008 (Atkinson, 1989; Gauthier & Rose,

								2002); COSEWIC, 2010
Redfish spp.	rigidity	larv.disp	1	3	3	3	3	Wourms, 1991; Pikanowski et al., 1999; COSEWIC, 2010
Redfish spp.	rigidity	pop.stat	3	5	15	5	15	COSEWIC, 2012
Redfish spp.	rigidity	resilience	2	5	10	5	10	Wourms, 1991;Musick, 1999; Pikanowski et al., 1999; Froese & Pauly, 2000; Sevigny et al., 2008; COSEWIC, 2010
Redfish spp.	sensitivity	diet.spec	1	2	2	2	2	Konchina, 1986; Pikanowski et al., 1999 (Steele, 1957; Yanulov, 1962; Marak, 1973; Sherman et al., 1984; Fortier et al., 1992; Anderson, 1994; Sameoto et al., 1994); Bowman et al., 2000
Redfish spp.	sensitivity	hab.spec	2	3	6	3	6	Hureau & Litvinenko, 1986; Scott & Scott, 1988; Campana et al., 1990; Wourms, 1991; Pikanowski et al., 1999 (Templeman, 1959; Kelly & Barker, 1961; Klein-MacPhee & Collette, in prep.; Scott, 1982; Shepard et al., 1986); Sevigny et al., 2008; COSEWIC, 2010
Redfish spp.	sensitivity	lat.range	2	2.45	4.9	2.45	4.9	iobis.org
Sandlance spp.	exposure	hab.avail	3	2.93	25.153757	4.45	88.121125	Shackell et al., 2014; OBIS-Canada
Sandlance spp.	exposure	larval.window	2	1	1	4	16	Buckley et al., 1984; Smigielski et al., 1984; Shackell & Frank,

								2000
Sandlance spp.	exposure	spawn.window	1	1	1	1	1	Smigielski et al., 1984
Sandlance spp.	rigidity	ad.mobility	2	2	4	2	4	Bigelow & Schroeder, 1953; Winslade, 1974; Nizinski et al., 1990; Nelson & Ross, 1991
Sandlance spp.	rigidity	larv.disp	1	4	4	4	4	Bigelow & Schroeder, 1953; Scott, 1972; Smigielski et al., 1984; Nizinski et al., 1990; Nelson & Ross, 1991
Sandlance spp.	rigidity	pop.stat	3	1	3	1	3	COSEWIC, 2012
Sandlance spp.	rigidity	resilience	2	2.3	4.6	2.3	4.6	Bigelow & Schroeder, 1953; Scott & Scott, 1988; Nizinski et al., 1990; Musick, 1999; Froese & Pauly, 2000
Sandlance spp.	sensitivity	diet.spec	1	3	3	3	3	Scott & Scott, 1988; Nizinski et al., 1990; Bowman et al., 2000; Coad & Reist, 2004
Sandlance spp.	sensitivity	hab.spec	2	4	8	4	8	Bigelow & Schroeder, 1953; Winters, 1983; Nizinski et al., 1990; Coad & Reist, 2004
Sandlance spp.	sensitivity	lat.range	2	3.95	7.9	3.95	7.9	iobis.org
Smooth Skate	exposure	hab.avail	3	1.64	4.410944	3.41	39.651821	Shackell et al., 2014; OBIS-Canada
Smooth Skate	exposure	larval.window	2	3	9	3	9	n/a
Smooth Skate	exposure	spawn.window	1	3	3	3	3	n/a
Smooth Skate	rigidity	ad.mobility	2	3	6	3	6	Bigelow & Schroeder, 1953; Walker et al., 1997; McEachran & Dunn, 1998; Packer et al., 2003; Carrier et al., 2010
Smooth	rigidity	larv.disp	1	4	4	4	4	Bigelow &

Skate								Schroeder, 1953; Breder & Rosen, 1966; McEachran & Dunn, 1998; Packer et al., 2003; McPhie & Campana, 2009; Carrier et al., 2010
Smooth Skate	rigidity	pop.stat	3	3	9	3	9	COSEWIC, 2012
Smooth Skate	rigidity	resilience	2	3.6	7.2	3.6	7.2	Breder & Rosen, 1966; Walker et al., 1997; McEachran & Dunn, 1998; Musick, 1999; Froese & pauly, 2000; Packer et al., 2003; McPhie & Campana, 2009; Carrier et al., 2010; Simon et al., 2012
Smooth Skate	sensitivity	diet.spec	1	1	1	1	1	Bowman et al., 2000; Packer et al., 2003; Carrier et al., 2010
Smooth Skate	sensitivity	hab.spec	2	2	4	2	4	Bigelow & Schroeder, 1953; Packer et al., 2003; Carrier et al., 2010
Smooth Skate	sensitivity	lat.range	2	3.54	7.08	3.54	7.08	iobis.org
Snow Crab	exposure	hab.avail	3	1.5	3.375	3.25	34.328125	Shackell et al., 2014; OBIS-Canada
Snow Crab	exposure	larval.window	2	4.5	20.25	5	25	Tremblay, 1986; Orensanz & Gallucci, 1988; Webb et al., 2006; Kuhn & Choi, 2011
Snow Crab	exposure	spawn.window	1	1	1	1	1	Tremblay, 1986; Webb et al., 2007
Snow Crab	rigidity	ad.mobility	2	3	6	3	6	Tremblay, 1986 (Hooper, 1986; Ennis et al., 1990; Sainte-Marie & Hazel, 1992; Dufour, 1988; Lefebvre & Brethes, 1991); Orensanz et al., 2005

Snow Crab	rigidity	larv.disp	1	2	2	2	2	Orensanz & Gallucci, 1988; Elner & Beninger, 1995; Conan et al., 1996; Bradbury & Snelgrove, 2001;Largier, 2003; Ernst et al., 2012
Snow Crab	rigidity	pop.stat	3	2	6	2	6	COSEWIC, 2012
Snow Crab	rigidity	resilience	2	3.6	7.2	3.6	7.2	Orensanz & Gallucc, 1988; Hines, 1991; Elner & Beninger, 1995; Conan et al., 1996; Musick, 1999; DFO, 2012; Ernst et al., 2012
Snow Crab	sensitivity	diet.spec	1	1	1	1	1	Wieczorek & Hooper, 1995; Lovrich & Sainte-Marie, 1997; Rouleau et al., 2000; Squires & Dawe, 2002
Snow Crab	sensitivity	hab.spec	2	2	4	2	4	Tremblay, 1986 (Elner, 1982; Dufour, 1988); Pinfold, 2006; DFO 2012
Snow Crab	sensitivity	lat.range	2	4.35	8.7	4.35	8.7	iobis.org
Spiny Dogfish	exposure	hab.avail	3	1	1	1	1	Shackell et al., 2014; OBIS-Canada
Spiny Dogfish	exposure	larval.window	2	3	9	3	9	Breder & Rosen, 1966; Compagno, 1984; McMillan & Morse, 1999; Muus & Nielsen, 1999; Fordham et al., 2006; Demirhan & Seyhan, 2007; Carrier et al., 2012
Spiny Dogfish	exposure	spawn.window	1	3	3	3	3	n/a
Spiny Dogfish	rigidity	ad.mobility	2	1	2	1	2	Aasen, 1960; Aasen, 1962; Compagno, 1984; McMillan & Morse, 1999; Muus & Nielsen, 1999; Fordham et al., 2006; Carrier et al., 2010

Spiny Dogfish	rigidity	larv.disp	1	1	1	1	1	Breder & Rosen, 1966; Compagno, 1984; McMillan & Morse, 1999; Muus & Nielsen, 1999; Fordham et al., 2006; Demirhan & Seyhan, 2007; Carrier et al., 2011
Spiny Dogfish	rigidity	pop.stat	3	4	12	4	12	COSEWIC, 2012
Spiny Dogfish	rigidity	resilience	2	5	10	5	10	Breder & Rosen, 1966; Compagno, 1984; Saunders & McFarlane, 1993; McMillan & Morse, 1999; Musick, 1999; Muus & Nielsen, 1999; Froese & Pauly, 2000; Pawson & Ellis, 2005; Fordham et al., 2006; Demirhan & Seyhan, 2007; Carrier et al., 2011
Spiny Dogfish	sensitivity	diet.spec	1	1	1	1	1	Jones & Geen, 1977; Ebert et al., 1992; Fujita et al., 1995; Ellis et al., 1996; Cortes, 1999; Bowman et al., 2000; Demirhan & Seyhan, 2007; ICES, 2012
Spiny Dogfish	sensitivity	hab.spec	2	1	2	1	2	Aasen, 1960; Aasen, 1962; Compagno, 1984; McMillan & Morse, 1999; Muus & Nielsen, 1999; Fordham et al., 2006; Carrier et al., 2010
Spiny Dogfish	sensitivity	lat.range	2	3.2	6.4	3.2	6.4	iobis.org
Thorny Skate	exposure	hab.avail	3	1.24	1.906624	2.17	10.218313	Shackell et al., 2014; OBIS-Canada
Thorny Skate	exposure	larval.window	2	3	9	3	9	n/a
Thorny	exposure	spawn.window	1	3	3	3	3	n/a

Skate								
Thorny Skate	rigidity	ad.mobility	2	2	4	2	4	Stenmann & Burkel, 1984; Compagno et al., 1989; Simon & Frank, 1996; Walker et al., 1997; Kulka & Mowbray, 1998; Skjaeraasen & Bergstad, 2000;Packer et al., 2003; Carrier et al., 2010
Thorny Skate	rigidity	larv.disp	1	4	4	4	4	Breder & Rosen, 1966; Stehmann & Bürkel, 1984; Scott & Scott, 1988; Mcphie & Campana, 2009b; Carrier et al., 2011
Thorny Skate	rigidity	pop.stat	3	3	9	3	9	COSEWIC, 2012
Thorny Skate	rigidity	resilience	2	3.6	7.2	3.6	7.2	Breder & Rosen, 1966; Walker et al., 1997; McEachran & Dunn, 1998; Musick, 1999; Froese & Pauly, 2000; Packer et al., 2003; Mcphie & Campana, 2009a/b; Carrier et al., 2010; Simon et al., 2012
Thorny Skate	sensitivity	diet.spec	1	1	1	1	1	Templeman, 1982; Scott & Scott, 1988; Compagno et al., 1989; Bowman et al., 2000; Skjaeraasen & Bergstad, 2000; Packer et al., 2003; Roman et al., 2004; Dolgov, 2005; ICES, 2012
Thorny Skate	sensitivity	hab.spec	2	2	4	2	4	Stehmann & Bürkel, 1984; Scott & Scott, 1988; Compagno et al., 1989; Kulka & Mowbray, 1998; Skjaeraasen & Bergstad, 2000;

								Packer et al., 2003; Carrier et al., 2010
Thorny Skate	sensitivity	lat.range	2	2.6	5.2	2.6	5.2	iobis.org
Turbot	exposure	hab.avail	3	1.27	2.048383	2.55	16.581375	Shackell et al., 2014; OBIS-Canada
Turbot	exposure	larval.window	2	3	9	4	16	Shackell & Frank, 2000; Godo & Haug, 1989
Turbot	exposure	spawn.window	1	1	1	1	1	Jorgensen, 1997; Bowering & Nedreaas, 2000
Turbot	rigidity	ad.mobility	2	1	2	1	2	Jorgensen, 1997; Bowering & Nedreaas, 2000;
Turbot	rigidity	larv.disp	1	1	1	1	1	Nielsen, 1986; Godo & Haug, 1989; Morgan et al., 2003; Coad & Reist, 2004
Turbot	rigidity	pop.stat	3	2	6	2	6	COSEWIC, 2012
Turbot	rigidity	resilience	2	3.6	7.2	3.6	7.2	Chugunova, 1959; Bowering, 1983; Nielsen, 1986; Godo & Haug, 1989; Vis et al., 1997; Musick, 1999; Froese & Pauly, 2000; Morgan et al., 2003; Murua & Saborido-Rey, 2003; Cooper et al., 2007
Turbot	sensitivity	diet.spec	1	2	2	2	2	Hart, 1973; Smith et al., 1978; Yang & Livingston, 1988; Bowering & Lilly, 1992; Pedersen & Riget, 1992; Chuchukalo et al., 1998; Orlov & Moukhametov, 2004; Bowman et al., 2000
Turbot	sensitivity	hab.spec	2	2	4	2	4	Nielsen, 1986; Coad & Reist, 2004
Turbot	sensitivity	lat.range	2	2.25	4.5	2.25	4.5	iobis.org
White Hake	exposure	hab.avail	3	1	1	1	1	Shackell et al., 2014; OBIS-Canada
White	exposure	larval.window	2	1	1	1	1	Chang et al., 1999

Hake									
White Hake	exposure	spawn.window	1	3	3	3	3	n/a
White Hake	rigidity	ad.mobility	2	2	4	2	4	Fahay & Able, 1989; Cohen et al., 1990; Chang et al., 1999 (Musick, 1974; MacDonald et al., 1984; Chang, 1990; Comyns & Grant, 1993; Langton et al., 1994)
White Hake	rigidity	larv.disp	1	1	1	1	1	Fahay & Able, 1989; Cohen et al., 1990; Chang et al., 1999
White Hake	rigidity	pop.stat	3	4	12	4	12	COSEWIC, 2012
White Hake	rigidity	resilience	2	4	8	4	8	Fahay & Able, 1989; Cohen et al., 1990; Chang et al., 1999; Musick, 1999; Froese & Pauly, 2000
White Hake	sensitivity	diet.spec	1	1	1	1	1	Langton & Bowman, 1981;Chang et al., 1999; Bowman et al., 2000
White Hake	sensitivity	hab.spec	2	2	4	2	4	Cohen et al., 1990 ; Chang et al., 1999 (Scott, 1982; MacDonald et al., 1984)
White Hake	sensitivity	lat.range	2	4.15	8.3	4.15	8.3	iobis.org
Windowpane	exposure	hab.avail	3	2.53	16.194277	4.16	71.991296	Shackell et al., 2014; OBIS-Canada
Windowpane	exposure	larval.window	2	1	1	1.47	2.1609	Shackell & Frank, 2000; Chang et al., 1999
Windowpane	exposure	spawn.window	1	1	1	1	1	Chang et al., 1999 (Bigelow & Schroeder, 1953; Wheatland, 1956; Smith et al., 1975)
Windowpane	rigidity	ad.mobility	2	2	4	2	4	Chang et al., 1999
Windowpane	rigidity	larv.disp	1	1	1	1	1	Fahay, 1983; Chang et al., 1999 (Bigelow & Schroeder, 1953; Morse & Able,

								1995); Neuman & Able, 2003
Windowpane	rigidity	pop.stat	3	1	3	1	3	COSEWIC, 2012
Windowpane	rigidity	resilience	2	2.3	4.6	2.3	4.6	Robins & Ray, 1986; Chang et al., 1999
Windowpane	sensitivity	diet.spec	1	1	1	1	1	Chang et al., 1999 (Hacunda, 1981; Langton & Bowman, 1981); Bowman et al., 2000
Windowpane	sensitivity	hab.spec	2	3	6	3	6	Chang et al., 1999 (Howell & Simpson, 1994; Langton et al., 1994); Neuman & able, 2003
Windowpane	sensitivity	lat.range	2	3.1	6.2	3.1	6.2	iobis.org
Winter Flounder	exposure	hab.avail	3	1.05	1.157625	1.32	2.299968	Shackell et al., 2014; OBIS-Canada
Winter Flounder	exposure	larval.window	2	1	1	1	1	Shackell & Frank, 2000; Pereira et al., 1999 (Pearcy, 1962; Itzkowitz & Schubel, 1983; Allen, 1989; Scarlett, 1991; Locke & Courtenay, 1995)
Winter Flounder	exposure	spawn.window	1	1	1	1	1	Pereira et al., 1999 (Bigelow & Schroeder, 1953; Stoner et al., 1999)
Winter Flounder	rigidity	ad.mobility	2	2	4	2	4	Scott & Scott, 1988; Murdy et al., 1997; Pereira et al.,1999; Stoner et al., 2001
Winter Flounder	rigidity	larv.disp	1	4	4	4	4	Fahay, 1983; Scott & Scott, 1988; Pereira et al., 1999 (Perlmutter, 1947; Pearcy, 1962; Itzkowitz & Schubel, 1983; Crawford & Carey, 1985; Allen, 1989; Scarlett, 1991; Locke &

								Courtenay, 1995)
Winter Flounder	rigidity	pop.stat	3	2	6	2	6	COSEWIC, 2012
Winter Flounder	rigidity	resilience	2	2.3	4.6	2.3	4.6	Lux, 1973; Fahay, 1983; Pereira et al., 1999; Musick, 1999; Froese & Pauly, 2000; Murua & Saborido-Rey, 2003
Winter Flounder	sensitivity	diet.spec	1	1	1	1	1	Hacunda, 1981; Langton & Bowman, 1981; Pereira et al., 1999; Bowman et al., 2000
Winter Flounder	sensitivity	hab.spec	2	3	6	3	6	Murdy et al., 1997; Pereira et al., 1999; Stoner et al., 2001
Winter Flounder	sensitivity	lat.range	2	3.75	7.5	3.75	7.5	iobis.org
Winter Skate	exposure	hab.avail	3	1.52	3.511808	3.03	27.818127	Shackell et al., 2014; OBIS-Canada
Winter Skate	exposure	larval.window	2	3	9	3	9	n/a
Winter Skate	exposure	spawn.window	1	3	3	3	3	n/a
Winter Skate	rigidity	ad.mobility	2	2	4	2	4	Packer et al., 2003 (McEachran, 1973; McEachran & musick, 1975; Michalopoulos, 1990; McEachran, 2002); Reide, 2004
Winter Skate	rigidity	larv.disp	1	4	4	4	4	Breder & Rosen, 1966; Robins & Ray, 1986; Packer et al., 2003 (Vlaadykov, 1936; Bigelow & Schroeder, 1953; Scott & Scott, 1988; McEachran, 2002)
Winter Skate	rigidity	pop.stat	3	4	12	4	12	COSEWIC, 2012
Winter Skate	rigidity	resilience	2	5	10	5	10	Breder & Rosen, 1966; McEachran & Dunn, 1998; Packer et al., 2003 (Bigelow &

								Schroeder, 1953; McEachran, 1973; McEachran & Martin, 1977; Musick, 1999; Froese & pauly, 2000; Northeast Fisheries Science Center, 2000; Mcphie & Campana, 2009
Winter Skate	sensitivity	diet.spec	1	4	4	4	4	McEachran & Dunn, 1998; Bowman et al., 2000; Packer et al., 2003
Winter Skate	sensitivity	hab.spec	2	3	6	3	6	Breder & Rosen, 1966; Robins & Ray, 1986; Scott & Scott, 1988; Packer et al., 2003 (Bigelow & Schroeder, 1953; Tyler, 1971; Scott, 1982; Michalopoulos, 1990; Gottschall et al., 2000
Winter Skate	sensitivity	lat.range	2	3.65	7.3	3.65	7.3	iobis.org
Witch Flounder	exposure	hab.avail	3	1	1	1.01	1.030301	Shackell et al., 2014; OBIS-Canada
Witch Flounder	exposure	larval.window	2	1	1	1	1	Cargnelli et al., 1999
Witch Flounder	exposure	spawn.window	1	1	1	1	1	Cargnelli et al., 1999 (Bigelow & Schroeder)
Witch Flounder	rigidity	ad.mobility	2	3	6	3	6	Bowering & Brodie, 1991; Cargnelli et al., 1999 (Powles & kohler, 1970)
Witch Flounder	rigidity	larv.disp	1	2	2	2	2	Russell, 1976; Nielsen, 1986; Cargnelli et al., 1999 (Reid et al., 1999)
Witch Flounder	rigidity	pop.stat	3	2	6	2	6	COSEWIC, 2012
Witch Flounder	rigidity	resilience	2	3.6	7.2	3.6	7.2	Nielsen, 1986; McRuer et al., 1997; Cargnelli et al., 1999; Musick, 1999; Froese &

									Pauly, 2000
Witch Flounder	sensitivity	diet.spec	1	3	3	3	3	Langton & Bowman, 1981; Cargnelli et al., 1999 (Maurer & Bowman, 1975; Bowman & Michaels, 1984Collette & Klein-MacPhee, in prep.); Bowman et al., 2000
Witch Flounder	sensitivity	hab.spec	2	4	8	4	8	Cargnelli et al., 1999 (Powles & Kohler, 1970; Martin & Drewry, 1978; Scott, 1982; MacDonald et al., 1984; Auster et al., 1991)
Witch Flounder	sensitivity	lat.range	2	3.25	6.5	3.25	6.5	iobis.org
WSS Cod	exposure	hab.avail	3	1.37	2.571353	3.55	44.738875	Shackell pers comm.
WSS Cod	exposure	larval.window	2	1	1	1	1	Jordaan & Kling, 2003; Bogstad et al., 2013
WSS Cod	exposure	spawn.window	1	1	1	2.33	2.33	Galloway et al., 1998; Johansen and Krogh 1914; Valerio et al. 1992; Brander & Hurley, 1992
WSS Cod	rigidity	ad.mobility	2	2	4	2	4	Fahay et al., 1999; Jordaan & Kling, 2003; Drinkwater, 2005
WSS Cod	rigidity	larv.disp	1	2	2	2	2	Muus & Dahlstrom, 1974; Russell, 1976; Robb, 1981; Fahay et al., 1999
WSS Cod	rigidity	pop.stat	3	5	15	5	15	COSEWIC, 2012
WSS Cod	rigidity	resilience	2	2.3	4.6	2.3	4.6	Cohen et al., 1990; Fahay et al., 1999; Musick, 1999; Dutil & Brander, 2003
WSS Cod	sensitivity	diet.spec	1	1	1	1	1	Artz, 1978; Langton & Bowman, 1980; Hacunda, 1981; Armstrong, 1982; Berestovsky, 1989; dos Santos

								& Falk-Petersen, 1989; Nordeide & Fossa, 1992; Greenstreet, 1996; Bromley et al., 1997; Bowman et al., 2000; Herrmann, 2004; ICES, 2012
WSS Cod	sensitivity	hab.spec	2	1	2	1	2	Cohen et al., 1990; Fahay et al., 1999; Drinkwater, 2005
WSS Cod	sensitivity	lat.range	2	2.2	4.4	2.2	4.4	iobis.org
WSS Cusk	exposure	hab.avail	3	1	1	4.18	73.034632	Shackell et al., 2014; OBIS-Canada
WSS Cusk	exposure	larval.window	2	1	1	1	1	Shackell & Frank, 2000
WSS Cusk	exposure	spawn.window	1	3	3	3	3	n/a
WSS Cusk	rigidity	ad.mobility	2	4	8	4	8	Cohen et al., 1990; Bergstad, 1991; Riede, 2004
WSS Cusk	rigidity	larv.disp	1	2	2	2	2	Russell, 1976; Cohen et al., 1990
WSS Cusk	rigidity	pop.stat	3	5	15	5	15	COSEWIC, 2012
WSS Cusk	rigidity	resilience	2	3.6	7.2	3.6	7.2	Cohen et al., 1990; Musick, 1999
WSS Cusk	sensitivity	diet.spec	1	1	1	1	1	Langton & Bowman, 1980; Cohen et al., 1990; Bergstad, 1991; Bowman et al., 2000; ICES, 2012
WSS Cusk	sensitivity	hab.spec	2	2	4	2	4	Svetovidov, 1986; Cohen et al., 1990; Bergstad, 1991
WSS Cusk	sensitivity	lat.range	2	4.45	8.9	4.45	8.9	iobis.org
WSS Haddock	exposure	hab.avail	3	1.02	1.061208	1.83	6.128487	Shackell et al., 2014; OBIS-Canada
WSS Haddock	exposure	larval.window	2	1	1	2.14	4.5796	Laurence, 1978; Cargnelli et al., 1999; Martell et al., 2005; Neuheimer & Taggart, 2010; Bogstad et al., 2013
WSS Haddock	exposure	spawn.window	1	1	1	1	1	Page and Frank, 1989; Neuheimer & Taggart, 2010
WSS Haddock	rigidity	ad.mobility	2	2	4	2	4	Cohen et al., 1990; Cargnelli et al., 1999; DFO, 2002

WSS Haddock	rigidity	larv.disp	1	2	2	2	2	Russell, 1976; Hurley & Campana, 1989; Brander & Hurley, 1992; Cargnelli et al., 1999
WSS Haddock	rigidity	pop.stat	3	1	3	1	3	COSEWIC, 2012
WSS Haddock	rigidity	resilience	2	2.3	4.6	2.3	4.6	Muus & Dahlstrom, 1978; Cargnelli et al., 1999; Musick, 1999; Froese & Pauly, 2000; Neuheimer & Taggart, 2010
WSS Haddock	sensitivity	diet.spec	1	2	2	2	2	Marak, 1960; Muus & Dahlstrom, 1978; Langton & Bowman, 1980; Robb & Hislop, 1980; Robb, 1981; Daan, 1989; Greenstreet, 1996; Jiang & Jorgensen, 1996; Bromley et al., 1997; Cargnelli et al., 1999; ICES 2012
WSS Haddock	sensitivity	hab.spec	2	1	2	1	2	Scott, 1982; Scott & Scott, 1988; Hurley & Campana, 1989; Brander & Hurley, 1992: Cargnelli et al., 1999; DFO, 2002
WSS Haddock	sensitivity	lat.range	2	3.75	7.5	3.75	7.5	iobis.org
WSS Offshore Hake	exposure	hab.avail	3	1	1	4.6	97.336	Shackell et al., 2014; OBIS-Canada
WSS Offshore Hake	exposure	larval.window	2	1	1	1	1	Chang et al., 1999 (Reid et al., 1999)
WSS Offshore Hake	exposure	spawn.window	1	3	3	3	3	n/a
WSS Offshore Hake	rigidity	ad.mobility	2	3	6	3	6	Cohen et al., 1990; Chang et al., 1999 (Bigelow & Schroeder, 1955); Lloris et al., 2005

WSS Offshore Hake	rigidity	larv.disp	1	1	1	1	1	Chang et al., 1999 (Morse, 1989; Reid et al., 1999); Lloris et al., 2005
WSS Offshore Hake	rigidity	pop.stat	3	3	9	3	9	COSEWIC, 2012
WSS Offshore Hake	rigidity	resilience	2	2.3	4.6	2.3	4.6	Cohen et al., 1990; Lloris et al., 2005
WSS Offshore Hake	sensitivity	diet.spec	1	2	2	2	2	Langton & Bowman, 1980; Chang et al., 1999 (Bigelow & Schroeder, 1955; Langton & Bowman, 1980; Cohen et al., 1990); Bowman et al., 2000
WSS Offshore Hake	sensitivity	hab.spec	2	1	2	1	2	Cohen et al., 1990; Chang et al., 1999 (); Lloris et al., 2005
WSS Offshore Hake	sensitivity	lat.range	2	1	2	1	2	iobis.org
WSS Pollock	exposure	hab.avail	3	1	1	3.63	47.832147	Shackell et al., 2014; OBIS-Canada
WSS Pollock	exposure	larval.window	2	1	1	2.14	4.5796	Cargnelli et al., 1999 (Bigelow & Schroeder, 1953; Hardy, 1978)
WSS Pollock	exposure	spawn.window	1	1	1	1	1	Collette & Klein-MacPhee, in prep.
WSS Pollock	rigidity	ad.mobility	2	2	4	2	4	Cohen et al., 1990; Frimodt, 1995; Bromley et al., 1997; Cargnelli et al., 1999 (Collette & Klein-MacPhee, in prep.; Hardy, 1978; Ojeda and Dearborn 1990; Rangeley and Kramer 1995)
WSS Pollock	rigidity	larv.disp	1	1	1	1	1	Cargnelli et al., 1999 (Bigelow & Schroeder, 1953; Hardy, 1978)
WSS Pollock	rigidity	pop.stat	3	2	6	2	6	COSEWIC, 2012
WSS Pollock	rigidity	resilience	2	2.3	4.6	2.3	4.6	Cargnelli et al., 1999 (Steele,

								1963; Mayo, 1994); Musick, 1999; Froese & Pauly, 2000
WSS Pollock	sensitivity	diet.spec	1	4	4	4	4	Marak, 1960; Robb, 1981; Greenstreet, 1996; Bromley et al., 1997; Cargnelli et al., 1999; Bowman et al., 2000
WSS Pollock	sensitivity	hab.spec	2	2	4	2	4	Cohen et al., 1990; Frimodt, 1995; Cargnelli et al., 1999 (Hardy, 1978; Scott, 1982)
WSS Pollock	sensitivity	lat.range	2	3.8	7.6	3.8	7.6	iobis.org
WSS Red Hake	exposure	hab.avail	3	1	1	3.32	36.594368	Shackell et al., 2014; OBIS-Canada
WSS Red Hake	exposure	larval.window	2	1	1	1	1	Steimle et al., 2000
WSS Red Hake	exposure	spawn.window	1	1	1	1	1	Steimle et al., 1999 (Wilk et al., 1990)
WSS Red Hake	rigidity	ad.mobility	2	2	4	2	4	Cohen et al., 1990; Frimodt, 1995; Steimle et al., 1999 (Bigelow & Schroeder, 1953; Tyler, 1971; Murawski & Finn, 1988; Jury et al., 1994; Stone et al., 1994; Able & Fahay, 1998)
WSS Red Hake	rigidity	larv.disp	1	1	1	1	1	Scotton et al., 1973; Svetovidov, 1986; Steimle et al., 1999
WSS Red Hake	rigidity	pop.stat	3	2	6	2	6	COSEWIC, 2012
WSS Red Hake	rigidity	resilience	2	2.3	4.6	2.3	4.6	Scotton et al., 1973; Svetovidov, 1986; Frimodt, 1995;Musick, 1999; Steimle et al., 1999; Froese & Pauly, 2000
WSS Red Hake	sensitivity	diet.spec	1	1	1	1	1	Langton & Bowman, 1980; Hacunda, 1981; Steimle et al.,

									1999; Bowman et al., 2000
WSS Red Hake	sensitivity	hab.spec	2	3	6	3	6	Cohen et al., 1990; Steimle et al., 1999 (Collette & Klein-MacPhee, in prep.; Gottschall et al., in review; Reid et al., 1979)
WSS Red Hake	sensitivity	lat.range	2	4.1	8.2	4.1	8.2	iobis.org
WSS Silver Hake	exposure	hab.avail	3	1	1	1	1	Shackell et al., 2014; OBIS-Canada
WSS Silver Hake	exposure	larval.window	2	1	1	1	1	Morse et al., 1999
WSS Silver Hake	exposure	spawn.window	1	1	1	1	1	Morse et al., 1999 (Schroeder, 1953; Fahay, 1974)
WSS Silver Hake	rigidity	ad.mobility	2	2	4	2	4	Scott & Scott, 1988; Cohen et al., 1990; Hesler et al., 1995; Lloris et al., 2005
WSS Silver Hake	rigidity	larv.disp	1	1	1	1	1	Hesler et al., 1995; Morse et al., 1999; Lloris et al., 2005
WSS Silver Hake	rigidity	pop.stat	3	2	6	2	6	COSEWIC, 2012
WSS Silver Hake	rigidity	resilience	2	2.3	4.6	2.3	4.6	Vinogradov & Kozachenko, 1989; Cohen et al., 1990; Hesler et al., 1995; Morse et al., 1999; Musick, 1999; Froese & pauly, 2000; Lloris et al., 2005
WSS Silver Hake	sensitivity	diet.spec	1	2	2	2	2	Langton & Bowman, 1980; Bowman, 1984; Vinogradov & Kozachenko, 1989; Morse et al., 1999; Bowman et al., 2000
WSS Silver Hake	sensitivity	hab.spec	2	2	4	2	4	Cohen et al., 1990; Hesler et al., 1995; Morse et al., 1999; Lloris et al., 2005
WSS Silver Hake	sensitivity	lat.range	2	3.6	7.2	3.6	7.2	iobis.org

APPENDIX F: LITERATURE FOR SPECIES FACTOR SCORES

Much of the literature gathered to inform fish species factor scores were obtained via Fishbase.org (Froese and Pauly, 2000). Where Fishbase.org could not supply specific enough information or data, additional literature searches were conducted using Google Scholar (Google, 2014) and Web of Science (Thomson Reuters, 2014).

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All study species

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