RECONCILING ECOLOGICAL RISK FRAMEWORKS AND POLICY IMPLICATIONS AT MULTIPLE SCALES

by

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ABSTRACT

Maintaining biodiversity, ecosystem stability, and avoiding functional boundaries are essential for maintaining a functioning biosphere. In this thesis, I analyse the effectiveness of two recent biodiversity policy and management tools at global and regional scales. I find that the Planetary Boundary framework has rarely been used directly to guide international conservation policy, though it has achieved modest adoption at the national scale in Europe. An alternative framework for evaluating biodiversity loss thresholds at a regional scale is the International Union for the Conservation of Nature Red List of Ecosystems (RLE). I create a reproducible methodology for assessing offshore marine ecosystems. I then conduct the first application of the RLE assessment on the Eastern Scotian Shelf (ESS) to provide both a contemporary ecosystem risk assessment and projections from an ecosystem model for the next century under a high emissions scenario. These result in the ESS ecosystem being categorized as "Endangered."

LIST OF ABBREVIATIONS USED

ABBREVIATION	**DESCRIPTION**
CBD	Convention on Biological Diversity
CMIP6	Coupled Model Intercomparison Project Phase 6
CPR	Continuous Plankton Recorder
CR	Critically Endangered
DD	Data Deficient
DFO	Fisheries and Oceans Canada
EN	Endangered
ESM	Earth System Model
ESS	Eastern Scotian Shelf
EU	European Union
IPBES	International Policy Platform on Biodiversity and Ecosystem Services
IPSL-CM5A-LR	Institute Pierre Simon Laplace Model Climate Model 5A Low Resolution
IPCC	Intergovernmental Panel on Climate Change
IUCN	International Union for the Conservation of Nature
LC	Least Concern
MPA	Marine Protected Area
NAFO	Northwest Atlantic Fisheries Organization
RLE	Red List of Ecosystems
RLTS	Red List of Threatened Species
SPM	Summary for Policy Makers

SST	Sea Surface Temperature
VU	Vulnerable
WSS	Western Scotian Shelf
WWF	World Wildlife Fund

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CHAPTER 1

INTRODUCTION

1.1 THE IMPORTANCE OF BIODIVERSITY

Biodiversity is key to maintaining stable and resilient ecosystems, that reliably provide ecosystem goods and services (IPBES, 2019a). It is the foundation of our economies, livelihoods, food security and health (Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services [IPBES], 2019a; Secretariat of the Convention on Biological Diversity [CBD], 2020a; World Wildlife Fund [WWF], 2020). For example, over 2 billion people rely on wood-burning for their energy (IPBES, 2019a), roughly 70% of cancer treatments are natural or synthetic products inspired by nature (IPBES, 2019a) and millions of people rely on the ocean for protein and micronutrients (Hicks et al., 2019; Worm et al., 2006). Biodiversity is also central to cultural aspects of life such as spirituality, recreation, and artistic inspiration (Plieninger et al., 2013; IPBES, 2019a). More food and materials are being derived from the natural world and appropriated by humanity than ever before (IPBES, 2019a). However, this is increasingly at the expense of the Earth's ability to continue to provide these ecosystem goods and services in a consistent and reliable manner in the future (IPBES 2019a). Even with judicious land and sea-use planning, supporting the expanding human population will present an immense challenge (CBD, 2020a). While technological developments may be able to replace some of these services and products, the diversity of nature secures humanity's ability to adapt to an uncertain future (IPBES 2019a), and pushing beyond the boundary of its ability to sustain us presents an enormous risk (Steffen et al., 2015).

1.2 BIODIVERSITY CRISIS

Biodiversity is deteriorating at an accelerating rate worldwide (IPBES, 2019a). The global rate of species extinction is, conservatively, tens of times higher than the average rate over the past ten million years (IPBES, 2019a). Between 1970 and 2016 the average monitored wildlife population abundances declined by 68% (WWF, 2020). Roughly one million species are currently threatened with extinction (IPBES, 2019a). Habitat destruction (e.g. land clearing for agriculture) is the greatest threat to terrestrial biodiversity, while exploitation (commercial fishing) is the greatest current threat to the marine environment (IPBES, 2019a). Other anthropogenic threats to both environments include invasive species, pollution, and, increasingly, climate change (IPBES, 2019a). In addition to climate change being a driver of biodiversity loss, biodiversity plays an important role in climate regulation, and hence the loss of biodiversity contributes to climate change (e.g. deforestation emits greenhouse gases) (IPBES, 2019a). While these interwoven crises have been identified by the United Nations as existential threats to humanity, the biodiversity crisis receives significantly less political attention (IPBES, 2019a).

Like anthropogenic warming, the consequences of biodiversity loss can be abrupt and nonlinear (e.g. Burkett et al., 2005; Scheffer et al., 2009). For example, in response to nutrient-loading-induced biodiversity loss, compounded by stochastic events, coral reef ecosystems can shift to an algae-dominated alternative state instead of responding linearly to the changing conditions (Scheffer et al., 2001). As such, increasing biodiversity loss is potentially putting us into tipping point/boundary states where the loss of ecosystem function and services is rapid, ecosystem collapse is possible, and recovery is slow or absent. The biodiversity crisis can be halted, but it will require proactive environmental policies, among other solutions (IPBES, 2019a).

1.3 HISTORY AND CURRENT STATE OF BIODIVERSITY POLICY

Halting the ongoing degradation of biodiversity has been an objective of the international community for decades. In 1993, the preeminent biodiversity convention, the Convention on Biological Diversity, was ratified by 168 signatory countries and entered into force (CBD, 2020b). In 2002, Parties to the Convention on Biological Diversity committed to significantly slowing the rate of biodiversity loss over that decade – a commitment which was not achieved (Butchart et al., 2010). With this failure, renewed commitments were made to slow the degradation of and improve the status of biodiversity by 2020 (Tittensor et al., 2014). The mechanism for this was the Aichi targets, a set of 20 targets designed to broadly reduce anthropogenic pressures on biodiversity, improve its status, and build capacity globally (CBD, 2020a; Tittensor et al., 2014). While select Aichi targets were achieved (eg. Aichi target 11 - protect 17% of land and 10% of the ocean) in some countries, none of the Aichi targets were achieved by the 2020 deadline at a global scale (CBD, 2020a). Furthermore, 12 Aichi indicators show worsening trends (CBD, 2020a; Rounsevell et al., 2020). This failure can largely be attributed to a lack of coherence between biodiversity science and biodiversity policy (Rounsevell et al., 2020). As the international community agrees to determine the post-2020 targets, which will continue to 2050, we are at an inflection point for biodiversity policy. The failure to achieve the aforementioned targets has left little room for error or regression. A critical step on the pathway to a sustainable future is the examination of existing biodiversity policies to determine their current efficacy and how they might be altered and built upon (Burgass et al., 2020).

The biodiversity planetary boundary framework (Rockström et al., 2009) and the International Union for the Conservation of Nature (IUCN) Red List of Ecosystems (RLE) assessment protocol (Bland et al., 2016) are two contemporary biodiversity policy tools that are alike in their focus on defining thresholds of ecological collapse. The biodiversity planetary boundary outlines a single, though currently unknown, shared threshold of biodiversity loss, spanning all of Earth's ecosystems, beyond which the risk of ecological collapse – and potentially profound impacts on society – increases (Steffen et al., 2015). While this unifying threshold of biodiversity loss is conceptually simple, it is challenging to move from the global level to quantify the boundary at regionally manageable scales (Nykvist et al., 2013). The IUCN RLE, in contrast, provides a framework to define unique thresholds of collapse for individual ecosystems (Bland et al., 2016). The framework has proven useful as a tool for ecosystem risk assessment in terrestrial, freshwater and coastal ecosystems (Bland et al., 2019). However, the usefulness of this framework has not been shown for offshore marine ecosystems: the single assessment to date (Bland et al., 2018) omitted one of the five assessment criteria, suggesting that further exploration is needed to determine how effective the RLE is as a risk assessment framework for offshore ecosystems.

While the conceptual basis of the biodiversity planetary boundary framework and the IUCN RLE assessment protocol has been widely discussed (Boitani et al., 2015; Mace et al., 2014; Montoya et al., 2018a, 2018b; Rockström et al., 2018; Steffen et al., 2015), examining their application as policy and management tools remains relatively unrealized.

1.4 RESEARCH OBJECTIVES

This thesis examines the suitability of two contemporary biodiversity policy and management tools at global and regional scales, respectively: the biodiversity planetary boundary framework (Rockström et al., 2009) and IUCN RLE assessment protocol (Bland et al., 2016). In Chapter 2, I investigate the uptake of the biodiversity (biosphere integrity) planetary boundary into policy. Planetary boundary theory was designed as a framework to aid in a shift in societal governance and management (Steffen et al., 2015), yet to date no thorough investigation has been performed to see how quickly - or if - it has been incorporated into international and national environmental policy. I perform a comprehensive review of policy documents to examine the uptake of the biosphere integrity planetary boundary into environmental policy.

In Chapters 3 and 4, I focus on examining the efficacy of the RLE assessment for offshore marine ecosystems. While the RLE assessment is increasingly used to define the risk status of ecosystems globally, the use of the protocol has only been demonstrated for terrestrial and coastal ecosystems. Using the Eastern Scotian Shelf (ESS) ecosystem as a case study, I examine the suitability of the RLE assessment protocol in offshore marine ecosystems, and build on the work of Bland *et al.* (2018) to develop a reproducible framework for performing RLE assessments that takes into account the unique aspects of offshore marine ecosystems.

In Chapter 4, I build on Chapter 3 to project changes to the ESS ecosystem under the Intergovernmental Panel on Climate Change (IPCC) “business as usual” carbon emissions scenario. As climate change has been identified as one of the greatest threats to biodiversity (IPBES, 2019a, 2019b), it is critical that decision-makers anticipate the impact of climate change on ecosystems. While rarely addressed, the last criterion (E) of the RLE requires projections of the potential for ecosystem collapse in the future. I, therefore, used a marine ecosystem model to project ecosystem changes in the ESS into 2100, and determine whether completing this criterion, and thus the RLE assessment protocol, is feasible in offshore marine ecosystems.

This thesis concludes with Chapter 5, a discussion of the overall findings, and future research directions.

Note that while the general introduction (Chapter 1) and conclusion (Chapter 5) are written in the first person, the three data chapters (Chapters 2, 3 and 4) are written as manuscripts and therefore include the use of ‘we’ in acknowledgement of my co-authors.

CHAPTER 2

THE UPTAKE OF THE BIOSPHERE INTEGRITY PLANETARY BOUNDARY CONCEPT INTO NATIONAL AND INTERNATIONAL ENVIRONMENTAL POLICY[1]

2.1 ABSTRACT

The biosphere integrity planetary boundary was, at least partly, developed to aid policymakers in addressing the dangerous decline of Earth's biodiversity. However, just over a decade since its origination the extent and speed of its adoption as a policy tool remains unclear. Here, we review the uptake of the biosphere integrity boundary into environmental policy at national and international scales, to determine the rapidity at which it has become embedded. We analyzed environmental reports published since 2009 by national governments in Europe and North America, and international reports by global biodiversity conventions and bodies. Our study found that over the last decade the framework has been referenced relatively infrequently at the international scale, though seen greater uptake at national scales, particularly in Europe. Assessing whether this represents a rapid policy uptake remains challenging due to the paucity of comparable studies on rates for analogous concepts. However, our findings suggest that the biosphere integrity planetary boundary has become relatively quickly and increasingly embedded into some national policy.

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[1] Hurley, I., & Tittensor, D.P. (2020). The uptake of the biosphere integrity planetary boundary concept into national and international environmental policy. *Global Ecology and Conservation, 22,* e01029

2.2 INTRODUCTION

It has been ten years since the concept of “planetary boundaries”, was introduced by Rockström *et al.* (2009). Planetary boundaries demarcate a ‘safe operating space’ for humanity, beyond which the Earth system state is characterized by increased risk and a compromised ability to sustain human society (Mace et al., 2014; Steffen et al., 2015). Such boundaries have been identified for the following nine processes: biosphere integrity, climate change, ocean acidification, stratospheric ozone depletion, biogeochemical nitrogen and phosphorus cycling, global freshwater use, land system change, chemical pollution and atmospheric aerosol loading (Rockström et al., 2009) Steffen et al., 2015). While the planetary boundary concept has been criticized (Montoya et al., 2018a, 2018b), and arguably miscommunicated or misinterpreted (Rockström et al., 2018), it is a concept with strong traction in the scientific literature with Rockström’s original paper now cited over 8000 times in Google Scholar (as of March 2020). However, planetary boundary theory was designed as a framework to aid in a shift in societal governance and management (for more discussion of the usefulness of the boundary concept see Rockström *et al.*’s introductory paper (2009)).

Yet, over ten years since the biosphere integrity boundary concept was originated, no thorough investigation has been performed to see how quickly - or if - it has been incorporated into international and national environmental policy. For biodiversity policy, 2020 is a year of transition, representing the end-point of the internationally-agreed 2011-2020 Aichi Biodiversity Targets followed by an imminent move into the still-evolving post-2020 Global Biodiversity Framework (Mace et al., 2018). Given this, we conducted a comprehensive review to address the knowledge gap around how quickly and deeply the biosphere integrity boundary has become embedded into biodiversity policy, and what can be learnt about the process by which this has (or has not) happened.

2.3 METHODS

The biosphere integrity boundary thresholds the ability of ecosystems to continue to provide goods and services to human society, and the risk of these benefits being threatened due to biodiversity loss and (Mace et al., 2014; Rockström et al., 2009; Steffen et al., 2015). The potential for losing vital biodiversity has been highlighted in the recently released Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) global assessment summary for policy-makers (IPBES, 2019a, 2019b), and is a very topical issue in international environmental policy with the ongoing negotiations around the post-Aichi 2020 Global Biodiversity Framework. Given this, here we assess the uptake of the biosphere planetary boundary into both international and national environmental policy, by identifying and analyzing specific search terms (Table 2.1) in over 900 key policy documents released since the inception of the concept. Search terms for this study were reviewed in English, French and Spanish, though most policy reports were accompanied by an English translation. Critical analysis of the identified search terms was conducted to exclude reports that used one, or more, of the search terms in a context unrelated to the biosphere boundary concept. Whilst our study methodology cannot account for any influence the planetary boundary framework may have had in preparatory stages of policies (for example on the Sustainable Development Goals as per Saunders, 2015), if the boundary concept is adopted as a policy tool then it would be retained in the final public versions. Therefore, in this study we define ‘uptake’ as the inclusion of both the concept and its associated terminology in policy reports.

At the international scale, we reviewed all international environmental reports published after 2009 (the year of Rockström’s original paper) by the relevant major biodiversity bodies and conventions, namely: the International Union for the Conservation of Nature (IUCN), IPBES and the Convention on Biological Diversity (CBD).

We then performed a national policy review guided by our findings. Following suggestions that the planetary boundary framework has primarily been used in Europe (Keppner, 2017), we focused our national environmental policy review in scope to European and North American countries. We also examined Colombia’s national biodiversity-related reports, based on the findings of the international review. Using the same search terms (Table 2.1), we systematically reviewed reports pertaining to biodiversity published since 2009 on national-level Ministry of the Environment websites for these countries to assess the rate at which the biosphere integrity boundary concept was being incorporated into the biodiversity agenda.

2.4 RESULTS

Internationally, the biosphere integrity boundary has been referenced by prominent bodies since its inception, but only relatively infrequently. The IUCN, first established in 1948, is one of the leading biodiversity organisations operating at a global scale. In 2015 the IUCN wrote a 2017-2020 programme to integrate the Sustainable Development Goals into their vision that included reference to the boundary (IUCN, 2016). Specifically, the programme suggested that poverty will not be eradicated while preserving the environment if society does not transform to live within planetary boundaries. As the IUCN's current 2020 vision was published in 2008, prior to the initial introduction of the planetary boundary framework, there is no mention therein.

IPBES is a leading biodiversity science-policy platform established in 2012. Notably, in their first global assessment (IPBES, 2019b), the biosphere integrity planetary boundary was frequently referenced. Specifically, the assessment addressed the risk of transgressing the biosphere integrity planetary boundary, though it was also acknowledged that there remains some debate regarding the existence and location of planetary boundaries (IPBES, 2019b). However, the planetary boundary framework was not mentioned in the summary for policy makers (SPM) (IPBES, 2019a). This is noteworthy as the SPM may be regarded as the most prominent and influential component of the global assessment.

The Convention on Biological Diversity is the primary multilateral biodiversity treaty, entering into force in 1993. Post-2009, countries party to CBD were required to submit their 5th national reports in 2014, and the 6th national reports in 2018, though many have not met this deadline. Of the 200 nations that submitted 5th national reports only one, Colombia, referenced the boundary. Correspondence with the lead author uncovered that Colombia became aware of the framework through direct collaboration with the Stockholm Resilience Centre, the birthplace of the planetary boundary concept (L.F. Vidal, personal communication, January 22, 2019). Of the 75 countries that had submitted their 6th national reports at the time of analysis, only Switzerland and Sweden referenced the boundary (~3% of reports). However, the biosphere boundary, at least as measured through our search terms and language scope (Table 2.1), was not mentioned in any of the ~170 reviewed National Biodiversity Strategies and Action Plans or in the zero-draft of the post-2020 biodiversity agreement.

At the national scale, the uptake of the boundary appears more frequent, but of those we examined seems to be limited to European states. Based on evidence that the biosphere integrity boundary had been transgressed, the European Union (EU) General Action Programme 2020 prioritized increasing knowledge of and evidence for planetary boundaries (European Parliament and the Council of the European Union, 2013). The programme calls for investment into research to define planetary thresholds for all boundaries that have yet to be quantified. Since the release of this action programme, six European countries (~12%), namely Germany, Switzerland, Sweden, Slovenia, Hungary and the Czech Republic, have integrated the planetary boundary framework into their national environmental policy. Many of these policy reports state the need to live within planetary boundaries. Given this appropriation of the concept of absolute limits to human use of biodiversity into national policy, the biosphere integrity boundary appears to already be useful as a qualitative and/or communication tool. Quantifying it may increase its utility for environmental governance by providing clearer information on how to stay within such limits.

2.5 DISCUSSION

Our decadal policy analysis and review has found that, since its conception, the biosphere integrity boundary has been increasingly embedded into environmental policy, though still to a somewhat limited extent. However, evaluating whether this represents a relatively slow or rapid uptake remains frustratingly difficult since points of comparison are rare. Analyses of the rapidity of the uptake of environmental science concepts from their original description in the peer-reviewed literature and then into policy documents appear in very short supply. Perhaps one analogue was the 20-year lag between the Stockholm Environment Institute suggesting a maximum two-degree increase of warming above pre-industrial levels and the signing of the Cancun agreement, committing governments to the suggested cap on warming (Pearce, 2014). Although this is not a strict equivalence, as it is climate and not biodiversity related, and furthermore is an apex target in an international agreement rather than simply the integration of a concept into policy documents, it suggests that the planetary boundaries concept has been embedded relatively quickly, although the uptake remains relatively geographically restricted. However, when calculating this time lag it is important to consider that science is not developed in a vacuum, and in fact the planetary boundary framework was conceived amidst an ongoing discussion of limits to human growth that began in the 1960s when space exploration began to give humans perspective on the uniqueness of Earth (Owens, 2013, 2015) and the environmental movement began demanding that scientists project the potential impacts of biodiversity loss (Jasanoff, 2012; Warde et al., 2018). Nevertheless, this lack of comparatives indicate that such rate analyses are valuable, as they could ultimately assist with identifying the means, mechanisms, and levers, as well as their relative importance, in moving from science into environmental policy. We therefore suggest that more analyses of the rapidity, and routes, by which environmental science is translated into policy may assist with more effectively streamlining this process in the future.

However, our results also indicate that the biosphere boundary concept has not effectively reached countries outside of Europe. This may perhaps be due to the European origin of the framework, the continued debate around the existence of planetary boundaries (Montoya et al., 2018a, 2018b), or because the dissemination of such concepts relies heavily on personal communication and direct links between scientists and policy-makers. This latter is perhaps suggested by the fact that one of the few countries to embed the biosphere integrity boundary into its national report, Sweden, is also the location for the origination of the concept at the Stockholm Resilience Centre. Furthermore, the transmission of this concept to Colombia, the only country to embed it into their 5th national report, also appears to be via direct communication between the key scientists involved and national policymakers. Alternatively, or additionally, the concentration of uptake in Europe could also be due to the historical awareness of and experience in Europe of the consequences of biodiversity loss. Indeed, while today large-scale landscape change and clearing occurs largely in tropical regions, Europe had already deforested most of its forests by the Middle Ages to clear land for agricultural use (Goldewijk, 2001). Finally, it is possible that the concept informed the preparatory stages of policy reports in other regions, but that these ideas were not specifically present in the final products. Regardless, at the ten year mark the biosphere planetary boundary has become a relatively common term in European biodiversity reports. An interesting avenue for further exploration would be to compare the uptake into policy of the other eight planetary boundaries.

While challenging, a specific identification and quantification of the biosphere integrity boundary (i.e. putting numbers on the where the boundary lies) at policy-relevant scales would increase policy opportunities (Nykvist et al., 2013). However, the acknowledgement alone of an absolute limit to the amount of biodiversity that humans can impact appears a powerful communication and policy tool. As seen in the public response to the IPBES global assessment, the urgency of the biodiversity crisis is finally being acknowledged at an appropriate scale. This momentum may provide an opportunity for policymakers to actively use and apply the biosphere planetary boundary in their efforts to ensure societal sustainability. Our analysis suggests that, while still relatively early, this process has already begun.

Table 2.1: Search terms used to review the uptake of the biosphere integrity planetary boundary into international and national environmental policy.

Language of report	**Search terms**
English	planetary, boundary, boundaries, biosphere integrity, biodiversity, Rockström, Mace, Steffen
French	planetaire, frontiere, limites, biodiversite, Rockström, Mace, Steffen
Spanish	planetario, limite, frontera, limites, biodiversidad, Rockström, Mace, Steffen

CHAPTER 3

RED LIST OF ECOSYSTEMS RISK ASSESSMENT FOR THE EASTERN SCOTIAN SHELF ECOSYSTEM

3.1 INTRODUCTION

3.1.1 BIODIVERSITY LOSS

Human society depends vitally upon ecosystem goods and services, which are underpinned by resilient, biodiverse, and functional ecosystems (IPBES, 2019a). Numerous studies have shown that biodiversity loss has negative consequences for ecosystems (e.g. Craven et al., 2016; Tilman et al., 2002; Worm et al., 2006) and that biological diversity is a powerful predictor of ecosystem function (Petchey et al., 2004; Tilman et al., 2002). Yet, the Earth's biodiversity is increasingly compromised by anthropogenic impacts that include overexploitation, climate change, pollution, habitat destruction, and invasive species (IPBES, 2019a; Rockström et al., 2009; Steffen et al., 2015), among others. These impacts may also tip ecosystems into new and degraded states in which their functioning is compromised (e.g. Mace et al., 2014; also see Chapter 1).

This reality requires society to enact practical policies to prevent and mitigate biodiversity loss, and hence to ensure the continued provision of ecosystem services, as well as to minimize the chance of ecosystems breaching tipping points or boundaries. This requires analysis of how ecosystems and biodiversity have changed over time and an assessment of their likelihood of continued deterioration or changes in state. While biodiversity depletion is often synonymous with species loss, it is in fact much more multifaceted (and scale-dependent) and can also include dramatic changes in the relative abundance of species, in habitat availability or suitability, homogenization of species composition, and declines in genetic diversity (Mace et al., 2014). Any of these expressions of biodiversity loss can conceivably alter ecosystem functioning, even in the absence of species extirpation or extinction. The IUCN's (International Union for the Conservation of Nature) Red List Index (RLI) is one of the most prominent tools used to assess species-level biodiversity loss as measured through extinction risk globally (Butchart et al., 2006), but the magnitude and multiple dimensions of decline require numerous tools and frameworks operating at different scales to guide management and policy bodies. While numerous such metrics of change exist at the species level (e.g. extinction risk, changes in range size, and species composition) (WWF, 2020), there are fewer that operate and integrate at the ecosystem level, and fewer still analogous to the Red List of Threatened Species (RLTS) that measure 'risk' in terms of ecosystem collapse.

3.1.2 IUCN RED LIST OF ECOSYSTEMS

The IUCN Red List of Ecosystems (RLE) is a recent addition to the toolbox of biodiversity assessments that examines loss at a broader scale and recognizes the importance of ecosystems and ecosystem degradation, in contrast to the indices above, which are focussed solely at the species level. The RLE provides an opportunity to evaluate the comparative state of the world's ecosystems. While human activity is firmly understood as a root cause of deteriorating ecosystem services (IPBES, 2019a; Mace et al., 2014), triaging conservation efforts towards ecosystems that exhibit the highest risk could be a more effective use of resources. But to do this, the risk status of all ecosystems must be determined in a consistent manner. The RLE provides a new way to categorize the vulnerability of ecosystems in an approach analogous to the RLTS, with eight categories of ecosystem risk (Figure 3.1). The overarching goal of the RLE is to aid conservation and management bodies in preventing further biodiversity loss, and ultimately ecosystem collapse (transformation into a novel ecosystem), by identifying the ecosystems with the greatest risk of collapse (Keith et al., 2015). In order to best inform management bodies, the RLE aims to complete a global assessment of marine and terrestrial ecosystems by 2025 (Bland et al., 2016).

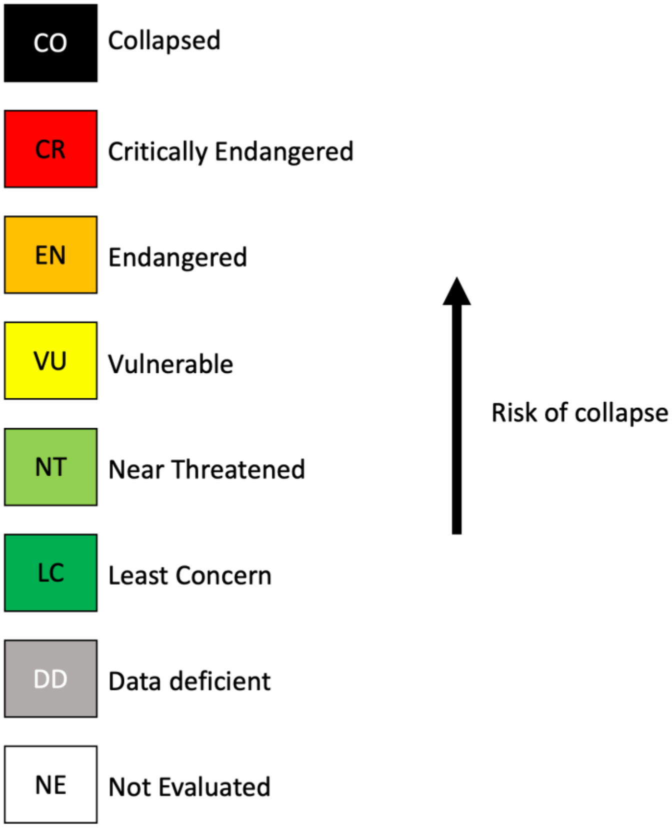

Figure 3.1: IUCN Red List of Ecosystem categories. Adapted from Bland *et al.* 2016.

Applying the RLE Assessment protocol involves five equally weighted criteria (A-E; Figure 3.2) that define ecosystem condition, where possible, by estimating: reductions in geographic distribution of the ecosystem (Criterion A); the current ecosystem size, measured as total area (km^2), against empirical spatial thresholds of risk (Criterion B); environmental degradation (Criterion C); declines in biotic ecosystem components (Criterion D), and lastly, the potential for ecosystem collapse in the next 100 years (Criterion E) (Bland et al., 2016; Figure 3.2; Table 3.1). Indicators for assessing the criteria are selected from a conceptual model of the ecosystem (Figure 3.6). Each criterion except B—which deals only in contemporary ecosystem size— is assigned a risk level based on the analysis of temporal trends for that specific ecosystem (Bland et al., 2016). The time periods across which risk is determined for criterion A, C and D include: the past 50 years, the next 50 years, and a historical range defined since 1750 A.D. (Figure 3.3). The requisite data for any historical analysis are rarely available and difficult to recover (e.g. alternative stable states), however, and although this category is perhaps the most valuable for establishing an accurate long-term ecosystem risk status, assessments including this period are rarely achieved. Criterion E deals with ecosystem-specific projections over the next 100 years (Bland et al., 2016). Finally, the RLE follows the precautionary principle: every ecosystem receives the categorization of the highest risk obtained for any of the criteria (Bland et al., 2016).

Criterion	A	B	C	D	E
	Reduction in distribution	**Restricted distribution**	**Environmental degradation**	**Disruption of biotic processes**	**Risk of future collapse**
Analysis	-Gather and process spatial data		-Select abiotic indicators and collapse thresholds. -Estimate relative severity and extent of degradation	-Select biotic indicators and collapse thresholds. -Estimate relative severity and extent of degradation	-Select abiotic and biotic indicators and collapse thresholds. -Implement appropriate ecosystem model.
Application	Extent over time: A1. Past 50 Years A2a. Next 50 years A2b. Any 50 year period A3. Since 1750	Current extent: B1. EOO B2. AOO B3. Number of locations	C1. Past 50 Years C2a. Next 50 years C2b. Any 50 year period C3. Since 1750	D1. Past 50 Years D2a. Next 50 years D2b. Any 50-year period D3. Since 1750	Project the ecosystem for 100 years
Thresholds	**A1, A2a and A2b:** CO = 100% decline CR. ≥ 80% decline EN. ≥ 50% decline VU. ≥ 30% decline **A3:** CO = 100% decline CR. ≥ 90% decline	See Table 3.1	**C1, C2a and C2b:** CO = 100% decline CR. ≥ 80% decline EN. ≥ 50% decline VU. ≥ 30% decline **C3:** CO = 100% decline CR. ≥ 90% decline EN. ≥ 70% decline VU. ≥ 50% decline	**D1, D2a and D2b:** CO = 100% decline CR. ≥ 80% decline EN. ≥ 50% decline VU. ≥ 30% decline **D3:** CO = 100% decline CR. ≥ 90% decline	CR. ≥ 50% within 50 years EN. ≥ 20% within 50 years VU. ≥ 10% within 100 years

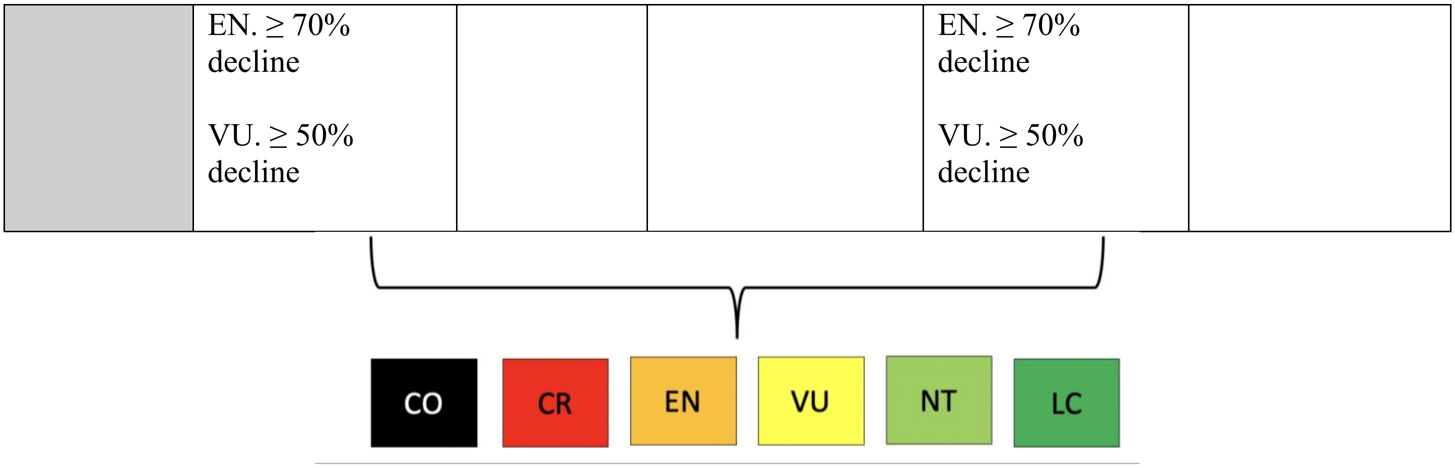

Figure 3.2: Assessment protocol for the Red List of Ecosystems. LC: Least Concern; VU: Vulnerable; EN: Endangered; CR: Critically Endangered. CO: Collapsed; EOO: Extent of occurrence; AOO: Area of occupancy. Figure adapted from Bland *et al.* (2016).

Table 3.1: Thresholds for risk categorization for criterion B. VU: Vulnerable; EN: Endangered; CR: Critically Endangered. EOO: Extent of occurrence; AOO: Area of occupancy. Table adapted from Bland *et al.* (2016).

Criterion B: restricted geographic distribution		**CR**	**EN**	**VU**
B1	Extent of ecosystem (Extent of all occurrences, EOO) is no larger than:	≤ 2,000 km^2	≤ 20,000 km^2	≤ 50,000 km^2
B2	The number of 10 × 10 km grid cells occupied (area of occupancy, AOO) is no more than:	≤ 2	≤ 20	≤ 50
B3	The number of threat-defined locations is very small AND prone to the effects of human activities or stochastic events within a very short time period, and thus capable of Collapse or becoming Critically Endangered (CR) within a very short time period (B3 can only lead to a listing as VU).			VU

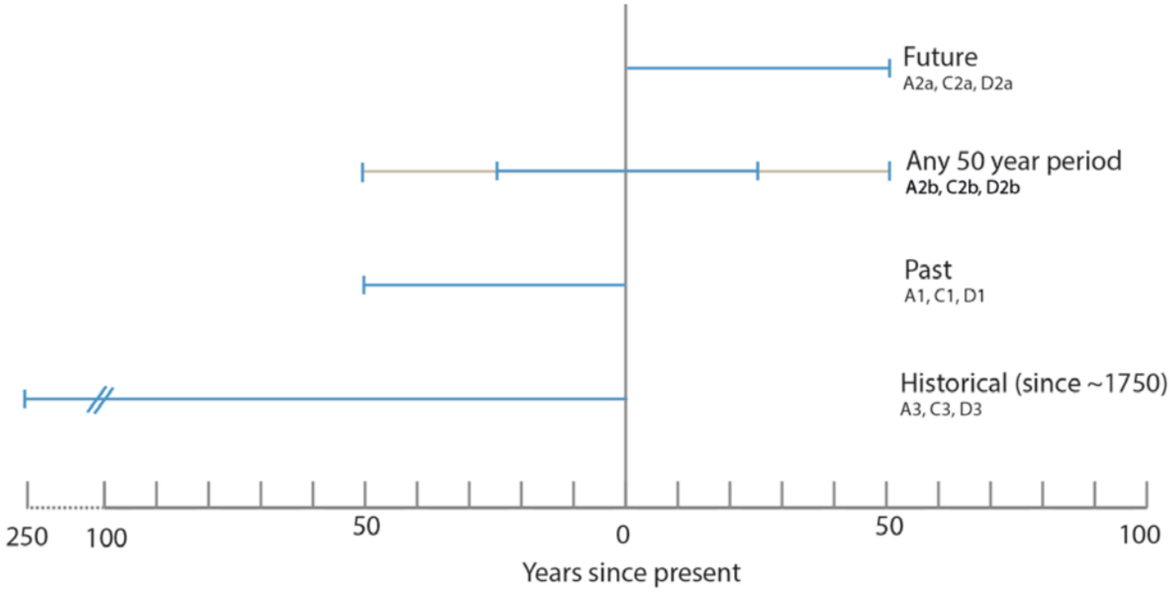

Figure 3.3: Time frame for assessment of criteria A, C and D. Figure from Bland *et al.* (2016).

To date over 2500 RLE assessments have been performed, yet only one offshore marine ecosystem (the southern Benguela; Bland et al., 2019) has been assessed. For our purposes offshore marine ecosystems are defined here as an oceanic environment, or a neritic environment not characterized by a dominant vegetation or habitat structure. However, the single offshore marine ecosystem assessed to date (Bland et al., 2018) omitted one of the five criteria, namely the potential for future ecosystem collapse (Criterion E); this is a frequent omission for both terrestrial and marine environments and may be due to the challenges of modelling and associated data requirements. Furthermore, this paucity suggests that RLE assessments of offshore marine ecosystems may be particularly challenging, given their unique characteristics such as the complexities of delineating ecosystem extent, and that further exploration is needed to determine how effective the RLE is as an appropriate and informative tool for such ecosystems.

In this chapter, we evaluate the RLE for the Eastern Scotian Shelf (ESS) using the 1970 ecosystem conditions as our baseline and a ~50-year time-series, of both physical and biological characteristics. The ESS is a relatively data rich offshore marine ecosystem located in the Canadian Exclusive Economic Zone off the Eastern coast of Nova Scotia (Canada) (Figure 3.4; Frank et al., 2006) that has undergone commercial exploitation over the past two centuries (Zwanenburg, 2000). In discussing our findings, we evaluate how effectively the RLE can be applied and may need modification, in offshore environments.

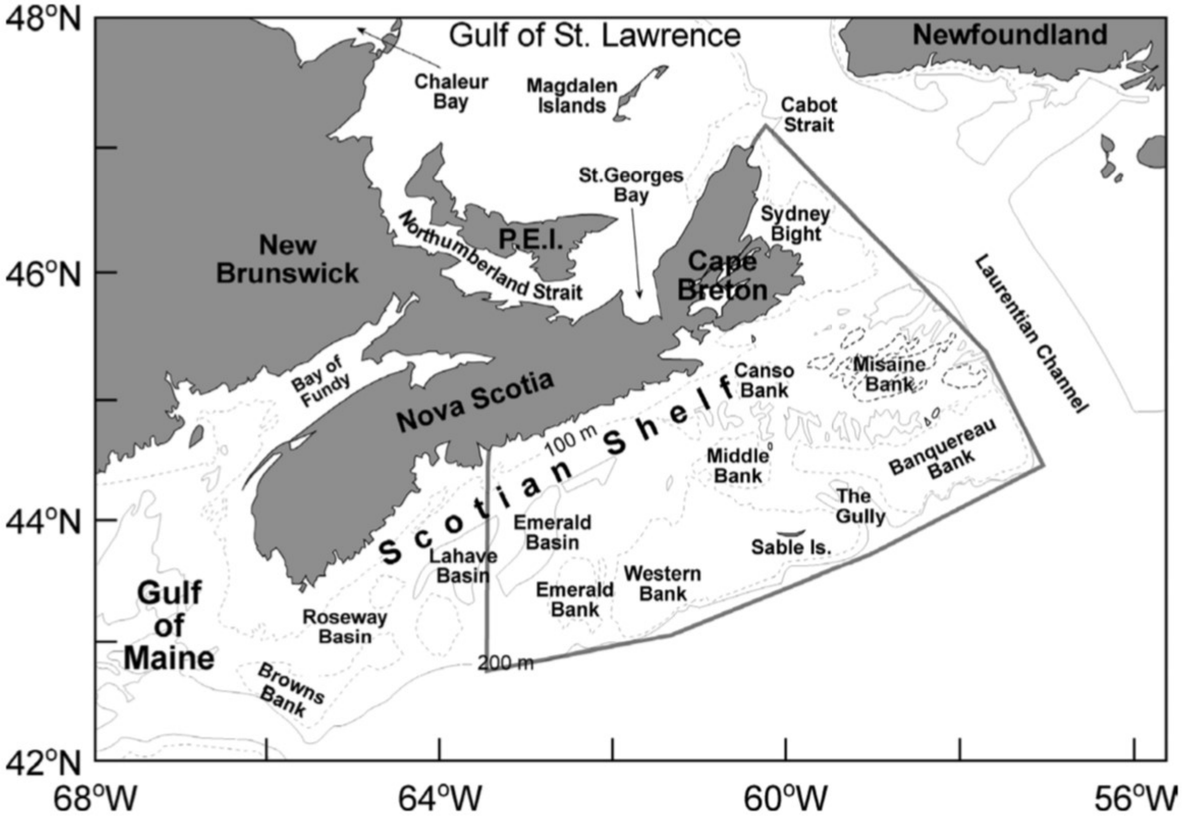

Figure 3.4: The bold line delineates the Eastern Scotian Shelf ecosystem. The study area excludes intertidal and coastal ecosystems. Figure from Frank *et al.* (2006).

3.1.3 EASTERN SCOTIAN SHELF ECOSYSTEM

The ESS is a biodiverse offshore marine ecosystem which comprises the Northwest Atlantic Fisheries Organization (NAFO) Division 4VW and covers ~108,000 km^2 (DFO, 2003; Frank et al., 2006; Figure 3.4). We only include offshore areas (depth >30m) as part of our assessment, thereby excluding inshore and intertidal regions. A series of gullies and channels that run between shallow offshore banks and inner basins are a distinctive feature of the ESS (DFO, 2003; Frank et al., 2006). The area includes the Gully marine protected area, which is recognized as an important site for cetaceans and deep-sea corals (Westhead et al., 2012), and the St. Ann's Bank marine protected area, a known migration corridor for many species (e.g. herring and blue whales) moving in and out of the Gulf of St. Lawrence (Hastings et al., 2010). The area also includes year-round fishery closures over Emerald Banks and Western Banks, which are important spawning areas for demersal species (DFO, 2003; Frank et al., 2006; Figure 3.4). While the boundary between the ESS and the neighbouring ecosystem, the Western Scotian Shelf (WSS) is not clearly defined by an obvious change in habitat structure, they have been managed by DFO as separate ecosystems since the 1970s, and notably the ESS is consistently colder and contains more offshore banks than the WSS, which may explain the somewhat different fish communities (see section 3.3.1; Stortini, 2020; Zwanenburg, 2000).

The ESS underwent a dramatic change in community composition following a substantial fisheries collapse almost three decades ago, caused in part by overfishing of demersal fish species (Frank et al., 2011; Sinclair et al., 2015). In the 1970s, demersal fish dominated the ESS ecosystem, including Atlantic cod, redfish, American plaice and silver hake (Frank et al., 2006). While less dominant in absolute numbers, the ESS provided refuge for small pelagic fish species such as Atlantic herring, capelin and sand lance (Frank et al., 2006). Following the collapse of the cod fishery in the early 1990s, a moratorium on directed fishing of cod and haddock was put in place in 1993 to help the stocks recover (DFO, 2003). However, following the implementation of the moratorium, major shifts in the abundance and relative dominance of characteristic species continued to be observed (Frank et al., 2005; Johanson et al., 2017). Notably, smaller pelagic fish have surpassed demersal species in absolute numbers (Frank et al., 2006) and prey on demersals at the juvenile stage (Minto & Worm, 2012). The structural shifts included a 900% increase in the biomass of forage fish (the primary prey of cod), declines in zooplankton, and an increase in phytoplankton (Frank et al., 2011). Paired with the moratoriums on groundfish, this has led to a significant increase in commercial fisheries' landings of pelagics and invertebrates (Figure 3.5, Frank et al., 2006). Grey seal (*Halichoerus gyprus*) populations have also increased exponentially since their near extirpation in the1960s (O'Boyle & Sinclair, 2012). Taken as a whole, the ESS has experienced conspicuous biological changes over the past 50 years, and in particular a significant shift from demersal dominated to pelagic-dominated in the 1990s.

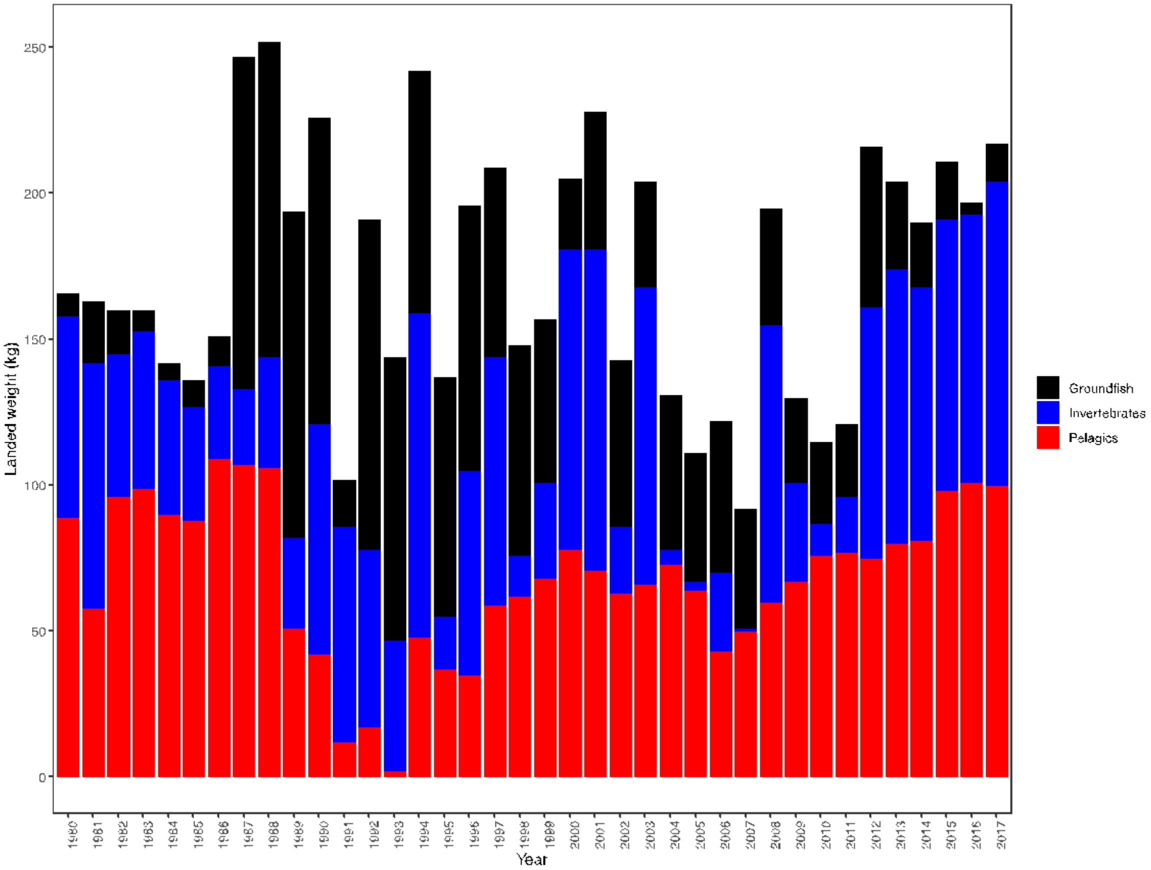

Figure 3.5: Landed weight of groundfish, pelagics and invertebrates on the Eastern Scotian Shelf from 1980-2017 (data provided by DFO).

A decade after the directed cod fishery closure in 1993 and the establishment of protected nurseries (Frank et al., 2006), ESS ecosystem monitoring in the mid-2000s indicated some biological changes that perhaps suggested a shift back towards a pre-collapsed state, such as the substantial increases in larger size classes of zooplankton (>2mm)— previously diminished as a result of the increasing abundance of their pelagic predators (e.g. herring). This trend was suggested to be linked to paired declines in pelagic fish (Frank et al., 2011). These changes meant that the ecosystem began to reflect the former trophic structure of the ESS, and perhaps the early stages of recovery (Frank et al., 2011). However, more recently, large zooplankton on the Scotian Shelf has shown significant declines, suggesting that the hopeful trends may be merely oscillations within the post-cod-collapse ecosystem, and not genuine recuperation (Bernier et al., 2018). Therefore, the recovery status of the ecosystem remains uncertain.

It is also unclear how the fisheries collapse, and perhaps limited potential recovery, might affect the RLE categorization and the probability of future ecosystem collapse. We assert that an RLE risk categorization of the ESS, rooted in observations that predate the cod collapse of the 1990s, could help guide policymakers to effectively consider the state of the ESS ecosystem and recovery. Any further recovery should also be reflected in a decrease in risk categorization under the RLE.

3.1.4 OBJECTIVES OF CHAPTER 3

The goal of this chapter was to assess the ESS ecosystem against the IUCN Red List of Ecosystems (RLE) protocol, with the broader purpose of evaluating the efficacy of the RLE for offshore marine ecosystems, and more specifically for substantially perturbed offshore marine ecosystems. In combination with the assessment of Criterion E in chapter 4, we aimed to determine the RLE categorisation for the ESS and provide it as a contribution to the global effort to categorise ecosystem vulnerability with these criteria.

3.2 METHODS

3.2.1 CONCEPTUAL MODEL OF THE ESS

A conceptual model of the ESS ecosystem was developed (Figure 3.6) based on the guidelines of the IUCN RLE (Bland et al., 2016), food web models of the ESS (Bundy, 2004), and select studies detailing the ESS ecosystem before and after the cod collapse (Bundy, 2004; Frank et al., 2005, 2006, 2011). While this model was necessarily simplified (i.e. does not include species groups such as seabirds and sharks), it does contain the key characteristic functional groups of the ecosystem, and therefore was sufficient for our purposes. The conceptual model was then used to guide the selection of indicators for the RLE assessment.

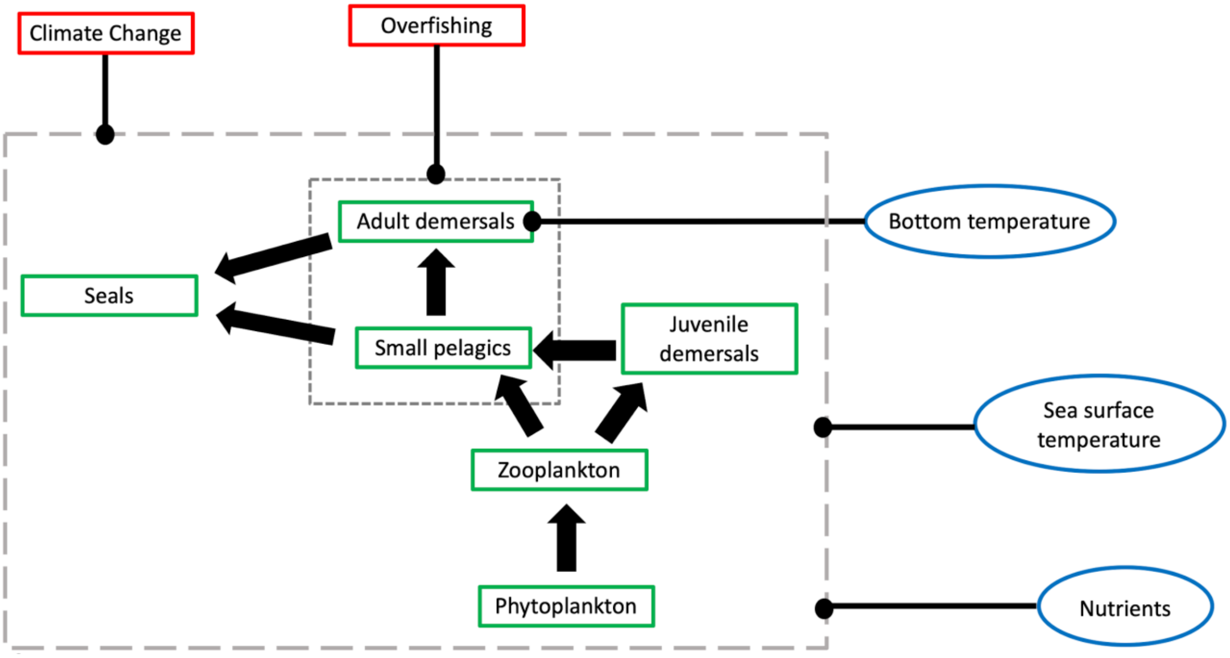

Figure 3.6: Conceptual model of the processes and interactions relevant to the Eastern Scotian Shelf ecosystem risk assessment. Blue ovals indicate environmental processes, green boxes biotic components, and red boxes threats to the ecosystem.

3.2.2 RED LIST OF ECOSYSTEMS CRITERIA

Criterion A: decline in spatial extent

Criterion A for the Red List of Ecosystems represents the risk associated with a decline in spatial extent (Bland et al., 2016). It is typically determined with a combination of remote sensing images and field observations, such as for the RLE evaluation of the tidal flats of the Yellow Sea and the RLE evaluation of Philippines fringe mangrove forests (Bland et al., 2016; Marshall et al., 2018; Murray et al., 2015; Table 3.2). This process is tractable when the ecosystem boundaries are clearly delineated and identifiable. However, examining spatial contraction is less straightforward for many offshore marine ecosystems, where boundaries are not clearly defined by a dominant vegetation or habitat structure. While some coastal and benthic ecosystems have identifiable habitat structures that aid in differentiating ecosystem borders (e.g. coral reefs), offshore marine ecosystems such as the ESS rarely do and often show functional and not spatial signs of decline (Bland et al., 2018).

One plausible approach would be to define the ESS through managerial rather than biological or environmental characteristics. Specifically, it could be defined by the NAFO division borders (4VW). However, any change in the extent of this ecosystem would then be driven by managerial and not biological characteristics and would not reflect a change in risk to the ecosystems. Instead, we define the ecosystem by the biological community, identified through its community composition as determined by an annual Fisheries and Oceans Canada (DFO) research vessel (RV) survey of the Scotian Shelf (for further survey details see DFO, 2013). To identify spatial changes in the ESS ecosystem over the period of observation (1970-2017), we used fuzzy clustering to group biological communities probabilistically and track changes in group coherence through time (Boyce et al., 2017).

To do so, the area of the RV survey, which encompasses both the ESS and the adjacent WSS, was divided into a 0.1 x 0.1 degree grid. The RV data was then divided into two eras based on the recognized community shift that occurred with the collapse of the cod fishery (<1993 and ≥1993). Because the RV survey data was not consistently resolved to the species level, the average abundance of genera per grid cell was used to create a matrix in which each row was a unique combination of cell and era. This matrix was then used to calculate the Bray-Curtis dissimilarity matrix to compare the ecological similarity of the grid cells in the same era. Fuzzy clustering, with two clusters, was then implemented on the Bray-Curtis matrix, which assigned each grid cell to one of the two clusters. Two clusters were chosen based on the distinct nature of the WSS and the ESS (Zwanenburg, 2000). Finally, the spatial distribution of the clusters was compared pre-1993 to post-1993 to determine if there had been a change in the spatial extent of the ESS. Following the method of Bland et al. (2018), we defined collapse as a total loss of the ecosystem ($0km^2$), which in this case would represent the WSS biological community across the entire ESS area. We were not able to quantify the future spatial decline of the ESS due to a lack of model projections of both genera abundance and spatial distribution at the highly resolved spatial and community level required for fuzzy clustering. Therefore, criterion A2 was listed as Data Deficient. Furthermore, as not uncommon for RLE assessments, we lacked the historical abundance data to assess criterion A3.

Table 3.2: An overview of approaches of previous IUCN Red List of Ecosystems assessments for each criterion. The approaches used in this study of the Eastern Scotian Shelf are provided for comparison. DD: Data deficient.

Criteria	Subcriteria	Common methods in existing RLE studies	Our approach
Criterion A	**A1**	Comparing time series of maps and/or remote sensing images	Applying fuzzy clustering to biological community spatial data (pre-1993 compared to 1993 and
	A2a		
	A2b		

	A3	(Keith et al., 2013; Murray et al., 2014)	after) to determine whether the ESS has experienced spatial decline.
Criterion B	**B1**	Determining the minimum convex polygon around the ecosystem (Bland et al., 2018; Keith et al., 2013)	Determining the minimum convex polygon around the ecosystem.
	B2	Calculating the number of 10 x 10 km grid cells (Bland et al., 2018; Keith et al., 2013)	Calculating the number of 10 x 10 km grid cells
	B3	Estimating number of areas within ecosystem and risk of threats in near future (~20 years) (Bland et al., 2018; Keith et al., 2013)	Estimating number of areas within ecosystem and risk of threats in near future (~20 years)
Criterion C	**C1**	Time series of sea level rise, temperature, retreat of ice mass, upwelling, sedimentation of coral reefs (Bland et al., 2016; Bland et al., 2018; Keith et al., 2013)	Time series of sea surface temperature and bottom temperature to compare to physiological temperature thresholds
	C2a	Time series of sea level rise, retreat of ice mass, upwelling, sedimentation of coral reefs (Bland et al., 2016, 2018)	Projected sea surface temperature and bottom temperature to compare to physiological temperature thresholds (see Chapter 4)
	C2b	Projections of sea level rise, retreat of ice mass, upwelling, sedimentation of coral reefs (Bland et al., 2016)	DD
	C3	Time series of sea level rise, retreat of ice mass, upwelling, sedimentation of coral reefs (Bland et al., 2016)	DD
Criterion D	**D1**	Times series of species richness, trophic diversity, abundance of key species, functional redundancy, ratios between key species groups (Bland et al., 2016; Bland et al., 2018; Keith et al., 2013)	Time series of abundance of key species, ratios of key functional groups
	D2a	Projections of species richness, trophic diversity, abundance of key species, functional redundancy (Bland et al., 2016)	DD

	D2b	Times series/ Projections of species richness, trophic diversity, abundance of key species, functional redundancy (Bland et al., 2016)	DD
	D3	Times series of species richness, trophic diversity, abundance of key species, functional redundancy (Bland et al., 2016; Keith et al., 2013).	DD
Criterion E		Ecosystem model (Bland et al., 2016)	Ecosystem model (see chapter 4)

Criterion B: restricted geographic distribution

Restricted geographic distribution, Criterion B, is examined through three subcriteria. Subcriteria B1 and B2 are used to evaluate the extent of the ecosystem (area of a minimum convex polygon) and the area of occupancy (number of occupied 10 x 10 km grid cells), respectively, against fixed area thresholds (Table 3.1). Criterion B3 determines the number of threat-based locations, where a threat-based location is defined as a distinct area where a single threat could cause a collapse in the near future (Table 3.1). For example, the Gnarled Mossy Cloud Forests of Australia are listed as “Vulnerable” under B3 as they are found in two distinct patches that are experiencing invasive rat predation on many canopy species, which could lead to ecosystem collapse within 20 years (Auld & Leishman, 2015).

To determine the extent of occurrence (criterion B1) we determined the area of a minimum convex polygon around the ESS ecosystem. To determine the extent of occupancy, a 10 x 10 km grid was superimposed over the ESS ecosystem area (Figure 3.4; criterion B2). Criterion B is used to identify ecosystems with small distributions and, therefore, susceptible to ecosystem collapse from a single threat or catastrophe (Bland et al., 2016). Given the large area of the ESS, and therefore the low risk under criterion B, we simply used managerial boundaries to assess B1 and B2. Numbers of locations within the ESS were identified, and then it was determined whether overfishing, a threat identified in the conceptual model (Figure 3.6), could cause ecosystem collapse, through dramatic declines or restructuring of the biological community, in the near future (~20 years as defined by previous assessments; Auld and Leishman 2015; Bland et al., 2018) by reviewing current fishing regulations (B3). While climate change was also identified as a threat to the ESS ecosystem (Figure 3.6), its potential impact was evaluated under Criterion E (see Chapter 4).

Selection of indicators for criteria C and D

Environmental and biotic degradation (criteria C and D) are determined by the relative severity of the decline in the chosen indicators for the period of assessment. Our protocol establishes the first five years of observations as a baseline (1970-1974). While the RLE recommends 50-year study periods, our study spanned 47 years (1970-2017) due to the availability of the data from the annual DFO RV survey of the Scotian Shelf, which was used to quantify the change over the recent past (subcriteria C1 and D1). Future functional changes (subcriteria C2 and D2) are projected using a separate ecosystem model, as described in Chapter 4. Like many RLE assessments, historical functional changes (subcriteria C3 and D3) were not quantified in our study due to the paucity of environmental and biological time series from the 1700s.

This study followed the protocol developed by Bland *et al.* (2018) in choosing indicators for criteria C and D. The protocol has four steps: (i) assess potential indicator relevance to ecosystem function; (ii) evaluate data availability for potential indicators; (iii) characterize trends relative to collapse; and (iv) identify suitable threshold values. A detailed description of the indicator selection process is outlined below.

Criterion C: degradation of the abiotic environment

Temperature constrains the development and growth rate of marine life (Barneche et al., 2019), and as such, represents an appropriate indicator under Criterion C. Here we used sea surface temperature (SST) and bottom temperature as indicators of environmental degradation (criterion C). Data for SST and bottom temperature were available from 1970 to 2017 from the annual DFO RV survey of the ESS. The RV survey data was consistently collected in July, but out of season observations also took place in select years (Figure A.1). To avoid intermittent seasonal temperature biases from wintertime extremes, observations from July were used exclusively. Nutrient concentrations (e.g. phosphate, nitrate and silicate) also affect marine life and therefore were also determined to be a relevant indicator of degradation of the abiotic environment (Moore et al., 2013). However, nutrient data were not available for much of our study period (Petrie et al., 1999). Therefore, SST and bottom temperature were the sole indicators examined under criterion C. Trends in abundance of phytoplankton and zooplankton are considered under biological indicators (see Criterion D).

Physiological tolerances result in species-specific thermal limits to survival. The thermal limits of fish species present in the RV survey data were obtained via Geo-located observations of species presence and temperature derived from the Aquamaps database (Kaschner et al., 2019; Table 3.3). The mean maximum and minimum thermal limits for demersal and pelagic species were used to create mean upper and lower bounds for the thermal ranges of both groups (Table 3.4). As pelagic fish live in the upper water column (Petrik et al., 2019), their thermal limits were compared to SST. Similarly, as demersal fish live near the seafloor (Petrik et al., 2019), their thermal limits were compared to bottom temperature. It was assumed that species could move to find a suitable temperature, leading us to compare pelagic and demersal thermal limits to mean observed SST and bottom temperature, respectively, instead of to the minimum and maximum SST and bottom temperature, respectively. If the mean SST and bottom temperature for the study period did not exceed the thermal limits of pelagics or demersals, criterion C would be listed as "Least Concern", otherwise it would be listed as "Collapsed" for pelagics and demersals independently.

Table 3.3: Complete list of demersal (D) and pelagic (P) species (as identified by Choi et al., 2004 from the annual DFO RV Survey of the Scotian Shelf) used in the analysis of criteria C and D. Thermal niches of the demersal and pelagic species used to estimate mean bottom temperature and sea surface temperature thresholds for the Eastern Scotian Shelf ecosystem. Minimum and maximum temperature data were obtained from Aquamaps observational data (Aquamaps, 2020).

	Common name	**Scientific name**	**Minimum (℃)**	**Maximum (℃)**
Demersals	American plaice	*Hippoglossoides platessoides*	-0.57	19.12
	Arctic eelpout	*Lycodes reticulatus*		
	Atlantic cod	*Gadus morhua*	-0.57	19.38

	Atlantic halibut	*Hippoglossus hippoglossus*	-0.92	18.98
	Atlantic sea poacher	*Agonus decagonus*	NA	NA
	Black belly rosefish	*Helicolenus dactylopterus*	1.77	30.93
	Black dogfish	*Centroscyllium fabricii*	0.13	28.3
	Brill/Windowpane	*Scophthalmus aquosus*	-1.61	26.49
	Cunner	*Tautogolabrus adspersus*	2.53	20.2
	Cusk	*Brosme brosme*	-0.57	18.98
	Eelpout	*Lycodes sp.*	NA	NA
	Fourbeard rockling	*Enchelyopus cimbrius*	-0.57	27.15
	Fourspot flounder	*Paralichthys oblongus*	NA	NA
	Haddock	*Melanogrammus aeglefinus*	-0.57	18.8
	Laval's eelpout	*Lycodes lavalaei*	-0.25	11.09
	Little skate	*Raja erinacea*	0.86	25.04
	Longfin hake	*Phycis chesteri*	2.66	27.48
	Longhorn sculpin	*Myoxocephalus octodecemspinosus*	1.68	18.1
	Marlin-spike grenadier	*Nezumia bairdii*	1	26.99
	Monkfish	*Lophius americanus*	-0.49	27.01
	Northern hagfish	*Myxine glutinosa*	-0.57	24.76
	Northern wolffish	*Anarhichas denticulatus*	-1.3	15.83
	Ocean pout	*Macrozoarces americanus*	-0.06	18.98
	Off-shore hake	*Merluccius albidus*	NA	NA
	Pollock	*Pollachius virens*	-0.57	23.47
	Redfish spp	*Sebastes spp*	-0.57	16.64

	Sea raven	*Hemitripterus americanus*	1.16	18.1
	Shortailed eelpout	*Lycodes vahlii*	-1.5	NA
	Silver hake	*Merluccius bilinearis*	0.53	26.95
	Smooth skate	*Raja senta*	0.13	19.38
	Snake blenny	*Lumpenus lumpretaeformis*	-0.57	19.71
	Spiny dogfish	*Squalus acanthias*	-0.57	28.51
	Spotted wolffish	*Anarhichas minor*	-0.57	15.57
	Squirrel or red hake	*Urophycis chuss*	1.18	25.62
	Striped atlantic wolffish	*Anarhichas lupus*	-1.57	20.03
	Thorny skate	*Raja radiata*	-1.36	25.04
	Turbot, Greenland halibut	*Reinhardtius hippoglossoides*	-1.22	20.2
	White barracudina	*Notolepis rissoi kroyeri*	NA	NA
	White hake	*Urophycis tenuis*	1.4	26.1
	Winter flounder	*Pseudopleuronectes americanus*	0.1	25.4
	Winter skate	*Raja ocellata*	1.75	23.37
	Witch flounder	*Glyptocephalus cynoglossus*	-0.24	20.2
	Wolf eelpout	*Lycenchelys verrillii*	-0.07	25.18
	Wrymouth	*Cryptacanthodes maculatus*	1.96	18.18
	Yellowtail flounder	*Limanda ferruginea*	1.55	18.98
Pelagics	Alewife	*Alosa pseudoharengus*	-0.14	24.15
	American sand lance	*Ammodytes americanus*	0.75	17.59
	American shad	*Alosa sapidissima*	-0.47	25.36
	Atlantic argentine	*Argentina silus*	NA	NA
	Atlantic herring	*Clupea harengus*	-0.57	25.4

	Atlantic mackerel	*Scomber scombrus*	-0.4	23.37
	Butterfish	*Peprilus triacanthus*	NA	NA
	Capelin	*Mallotus villosus*	-1.61	19.06
	Northern sand lance	*Ammodytes dubius*	-1.18	18.6

Table 3.4: Mean estimated thermal limits of demersals and pelagics in the ESS ecosystem.

	Minimum (°C)	**Maximum (°C)**
Demersals	0.1	22.1
Pelagics	-0.5	21.9

Criterion D: disruption of biological community

Again, following the protocol developed by Bland et al. (2018), we selected five indicators that represent characteristic functional groups of the ESS ecosystem in order to examine biotic decline (Criterion D): phytoplankton abundance, zooplankton abundance, survey-based demersal weight, survey-based ratios of demersal to pelagic weight, and grey seal abundance. Pelagic weight and abundance trends can be found in the appendix (Figure A.2; A.3). To determine relative changes while reducing interannual variability, the mean value of each indicator from 2013-2017 was compared to the mean value of the indicator from 1970-1975. In keeping with previous RLE assessments (Bland et al., 2018), a linear relationship was assumed between relative decline and ecosystem degradation for all indicators. As thresholds of collapse were uncertain, we conservatively set collapse thresholds for indicators at zero abundance or biomass, as previous applied in many RLE assessments (e.g Keith et al., 2013 and supplementary information therein; Murray et al., 2015).

Phytoplankton and zooplankton.

Phytoplankton Colour Index data from the Continuous Plankton Recorder (CPR) database were provided by DFO for the years 1970-2017 (with a data gap from 1977-1990). The Phytoplankton Colour Index is a visual proxy of cholorophyll based on the shade of green of the CPR filter cloth (Reid et al., 1998). While the phytoplankton colour index data is likely less accurate than a more precise tool (e.g. flow cytometry; Dubelaar and Jonker, 2000),, due to the paucity of phytoplankton sampling in the ESS ecosystem on the temporal and spatial scale required, the colour index was the best alternative.

Larger size class zooplankton (>2mm and hereafter large zooplankton) abundance data were also obtained from the CPR database for the years 1970-2017 (with a data gap from 1977-1990). Following the protocol of Frank *et al.* (2005), changes in the abundance of zooplankton on the ESS were examined through the summed abundance of three characteristic large zooplankton species: *Calanus finmarchicus, Calanus glacialis* and *Calanus hyperboreus.*

Grey seals

The abundance of grey seals was examined through modelled ESS grey seal abundance data, provided by DFO for the years 1970-2014 (for further model details see Hammill et al., 2017).

Demersals

Mean weight of demersal species (as identified by Choi et al. 2004, Table 3.3) per survey for each year (kg survey^{-1} year^{-1}) was calculated from the same 47-year data set of the annual DFO RV survey of the ESS that was used to obtain temperature data for criterion C (see above). Demersal abundance trends can be found in the appendix (Figure A.4).

Ratio of demersals to pelagics

While changes in abundance of the characteristic species groups of the ESS ecosystem are independently informative, it is instructive to examine the changing ratios of species groups. Such changes can signal community restructuring or ecosystem shifts. Using the same DFO survey described above, ratios of mean demersal to pelagic weight were examined by year (see figure A.5 for ratios of abundance). Species were placed in the demersal or pelagic groups based on Choi *et al.* (2004) (Table 3.3).

3.3 RESULTS

3.3.1 CRITERION A: DECLINE IN SPATIAL EXTENT

The WSS and ESS were distinctly identified and separated before 1993 by the fuzzy clustering algorithms of the genera assemblages (Figure 3.7), represented by blue and red grid cells respectively. While some overlap exists, red grid cells mostly (82% <1993 and 90% ≥1993) appear within the ESS and vice versa, and the two clusters largely distinguish each zone (Figure 3.7). Prior to 1993 the ESS community included 191 genera— 45 of which were unique to its cluster— and was dominated by flounders (*Hippoglossoides* and *Limanda*), cod (*Gadus*) and haddock (*Melanogrammus*). The WSS community included 182 genera and 51 unique to its cluster and was dominated by hakes (*Merluccius* and *Urophycis*), redfish (*Sebastes*), and haddock. Along the whole Scotian Shelf, the area occupied by the WSS and ESS species communities changed over the period examined. Using the total number of grid cells assigned to each cluster as a proxy for the spatial extent of that ecosystem, clear changes occurred after 1993. The WSS increased in area by 37%, from 572 grid cells to 784 grid cells in the cluster reconstruction after 1993 and expanded eastwards. The concomitant “loss” in the spatial extent of the ESS, at least within the area considered here, therefore results in it being defined as “Vulnerable” under criterion A of the IUCN RLE protocol (Table 3.5). However, it is worth noting that these results could also potentially be explained by species within the ESS community shifting northward in reaction to warming temperatures (Pinsky et al., 2020). Regardless of the cause, however, the composition of species on the ESS does appear to have changed over the past 50 years.

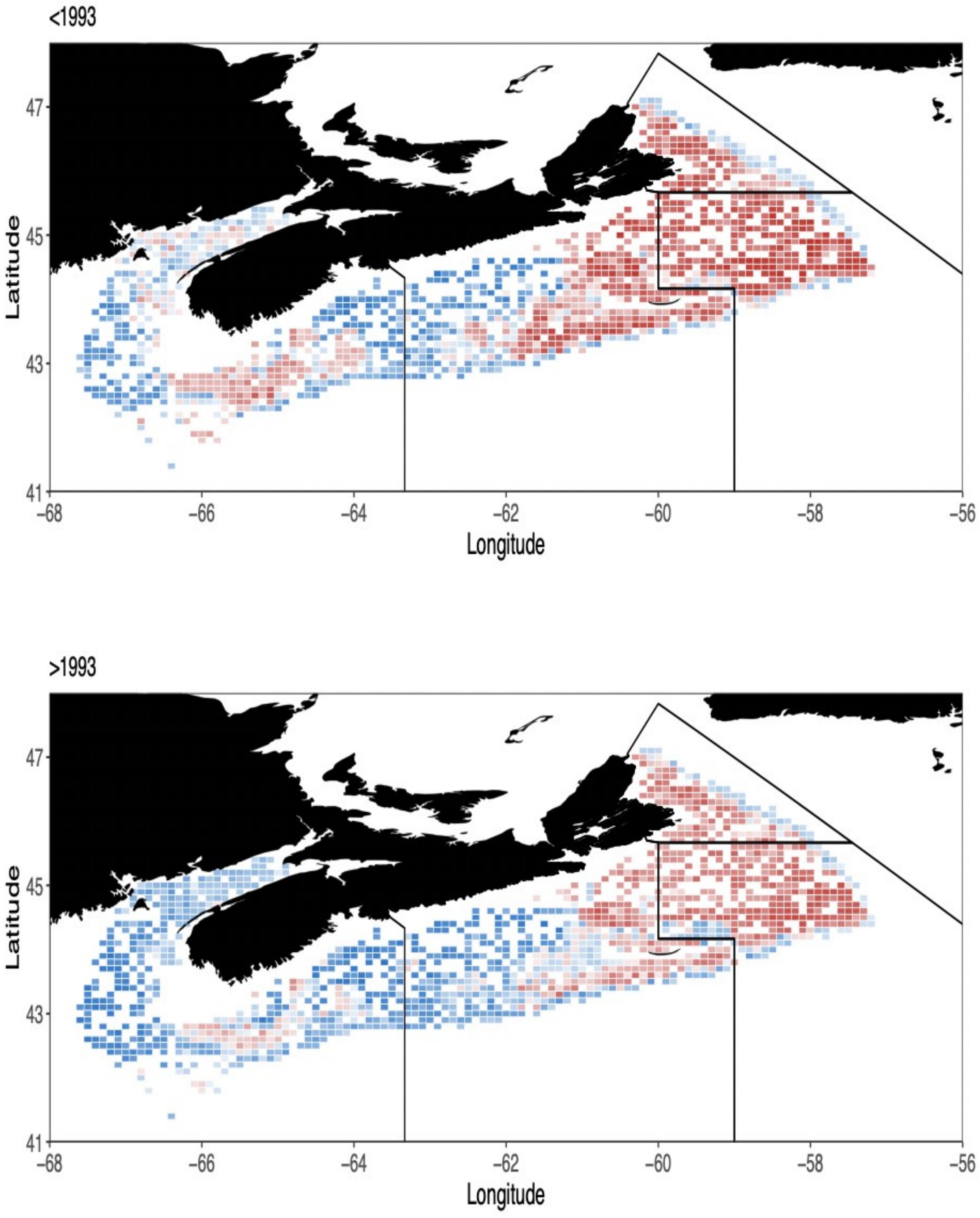

Figure 3.7: Spatial distribution of the species assemblage clusters from the fuzzy clustering algorithm. Colours depict cluster membership (blue is Western Scotian Shelf, and red Eastern Scotian Shelf), and transparency illustrates the probability of cluster membership: opaque grid cells represent high membership probability. The black borders delineate the following spatial areas: 4W, 4Vs and 4Vn.

3.3.2 CRITERION B: RESTRICTED GEOGRAPHIC DISTRIBUTION

The extent of occurrence of the ESS based on managerial boundaries is ~ 108,000 km^2 (Figure 3.8) and the area of occupancy is >5,000 grid cells (on a 10 x 10 km grid; Figure 3.8)— supporting a categorization of “Least Concern” under subcriteria B1 and B2 (Table 3.5).

In terms of subcriterion B3, the number of locations, while some ecosystems are clearly made up of distinct patches (e.g. seagrass meadows), it is unclear how patches would be defined for offshore marine ecosystems that are characteristically continuous. Therefore, the ESS ecosystem represents a single location, albeit with potentially heterogenous benthic habitat within, rather than multiple patches. While overfishing was identified as a potential threat to the ESS, strict fishing regulations have been in place since the cod collapse in the 1990s, namely the moratorium on directed fisheries of cod and haddock (Frank, 2005). Therefore, subcriterion B3 is categorized as “Least Concern” (Table 3.5) as, assuming that current management regulations and quotas remain, it is unlikely that overfishing will cause ecosystem collapse within 20 years.

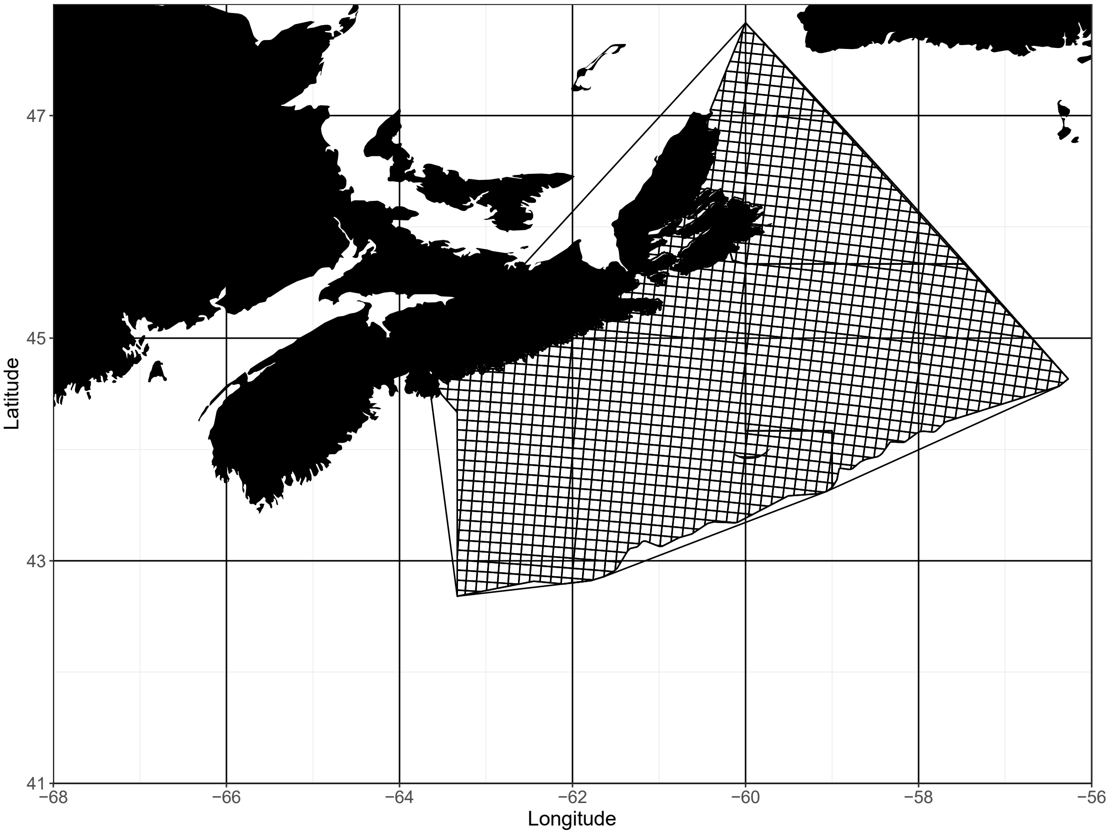

Figure 3.8: Spatial distribution of the Eastern Scotian Shelf and assessment of criterion B. The black line indicates the minimum convex polygon enclosing all ecosystem occurrences (extent of occurrence) and the grid cells indicate areas occupied at 10 x 10 km resolution (area of occupancy).

3.3.3 CRITERION C: ENVIRONMENTAL DEGRADATION

The annual mean SST during the period examined did not surpass the upper or lower threshold for the average physiological requirements of the pelagic species from 1970 to 2017 (Figure 3.9), and therefore the SST indicator was of "Least Concern." Similarly, the mean bottom temperature did not exceed the upper or lower thermal niche thresholds for the demersal species at any time, leading us to also categorize the bottom temperature indicator as "Least Concern" (Figure 3.9). From these results, we defined environmental degradation (criterion C) as "Least Concern" on the ESS (Table 3.5).

Figure 3.9: Minimum, maximum and mean SST and bottom temperature on the Eastern Scotian Shelf from 1970 to 2017. Purple dashed lines delineate the thresholds for pelagics (sea surface temperature) and demersals (bottom temperature) for mean temperature.

3.3.4 CRITERION D: DISRUPTION OF BIOTIC PROCESSES

Phytoplankton

Between 1970 and 2017, the five-year mean phytoplankton colour index increased by 300% (Figure 3.10). Assuming that the Continuous Plankton Recorder's colour index is an accurate estimate of phytoplankton abundance on *average*, under the RLE assessment protocol this qualifies as "Least Concern" (Figure 3.15).

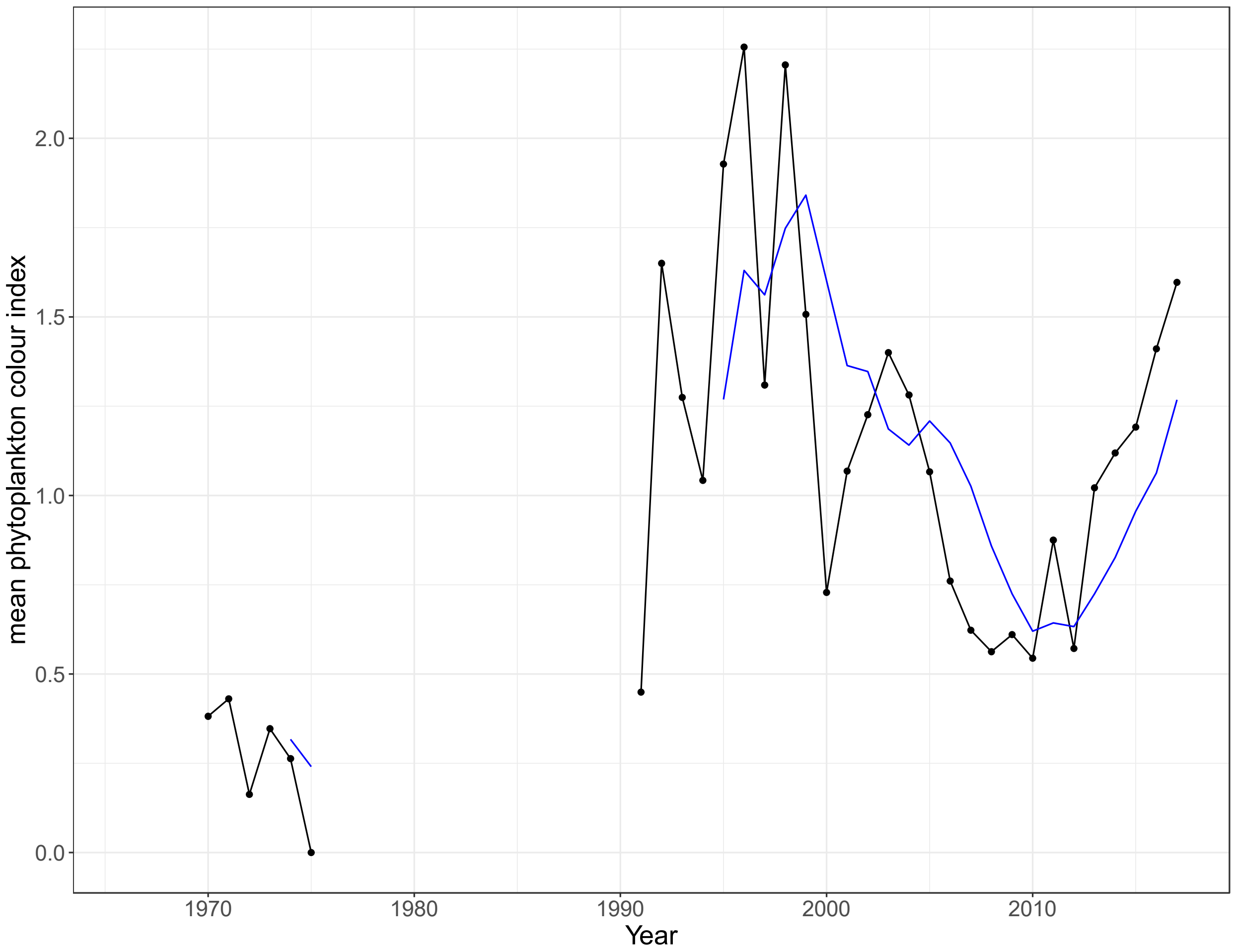

Figure 3.10: Temporal trend of the mean phytoplankton colour index on the Eastern Scotian Shelf from 1970 to 2017. Blue line represents the five-year moving average.

<u>Zooplankton</u>

Between 1970 and 2017, though mean zooplankton abundance does show substantial annual variability, the five-year mean of large zooplankton abundance decreased by 78% (Figure 3.11). This decline means that zooplankton abundance is classed as "Endangered" under the RLE assessment protocol (Figure 3.15). Furthermore, the degree of the decline resulted in the categorization of the zooplankton indicator at the extreme range of the "Endangered" category (Figure 3.15): if the zooplankton abundance declines an additional 2% in the future, it would be categorized as "Critically Endangered".

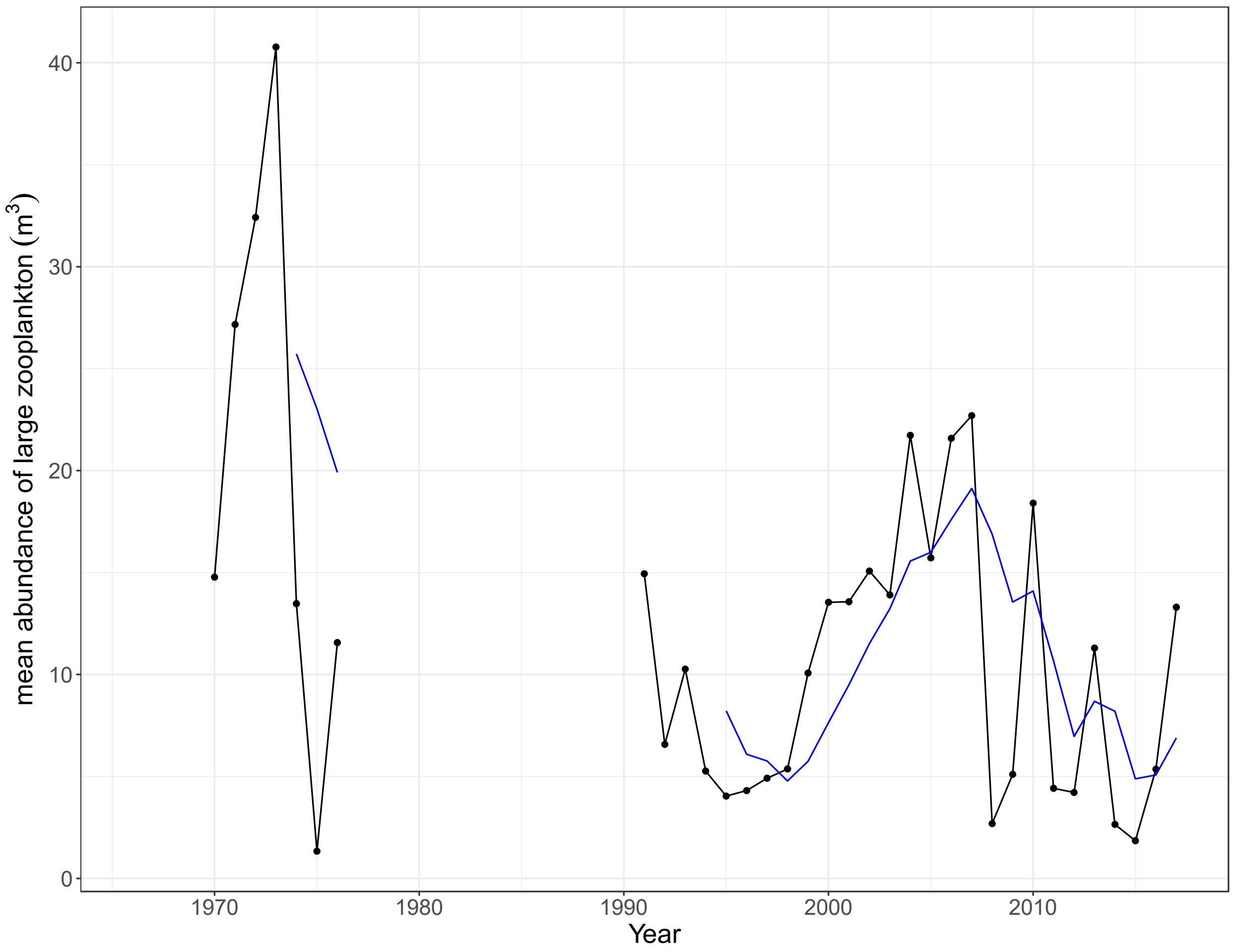

Figure 3.11: Temporal trend of mean abundance of large zooplankton (*Calanus finmarchicus, Calanus glacialis* and *Calanus hyperboreus*) from 1970 to 2017 in the Eastern Scotian Shelf. Blue line represents the five-year moving average.

Demersal changes

The five-year mean demersal weight (kg/survey) declined by 25.7% from 1970 to 2017 (Figure 3.12), placing it in the "Near Threatened" category (Figure 3.15). Starting in 1970, mean demersal weight (kg/survey) increased on average, and eventually peaked in 1983 (Figure 3.12). Since this maximum in 1983, the mean demersal weight (kg/survey) has steadily declined with its lowest weight recorded in 2015 (Figure 3.12).

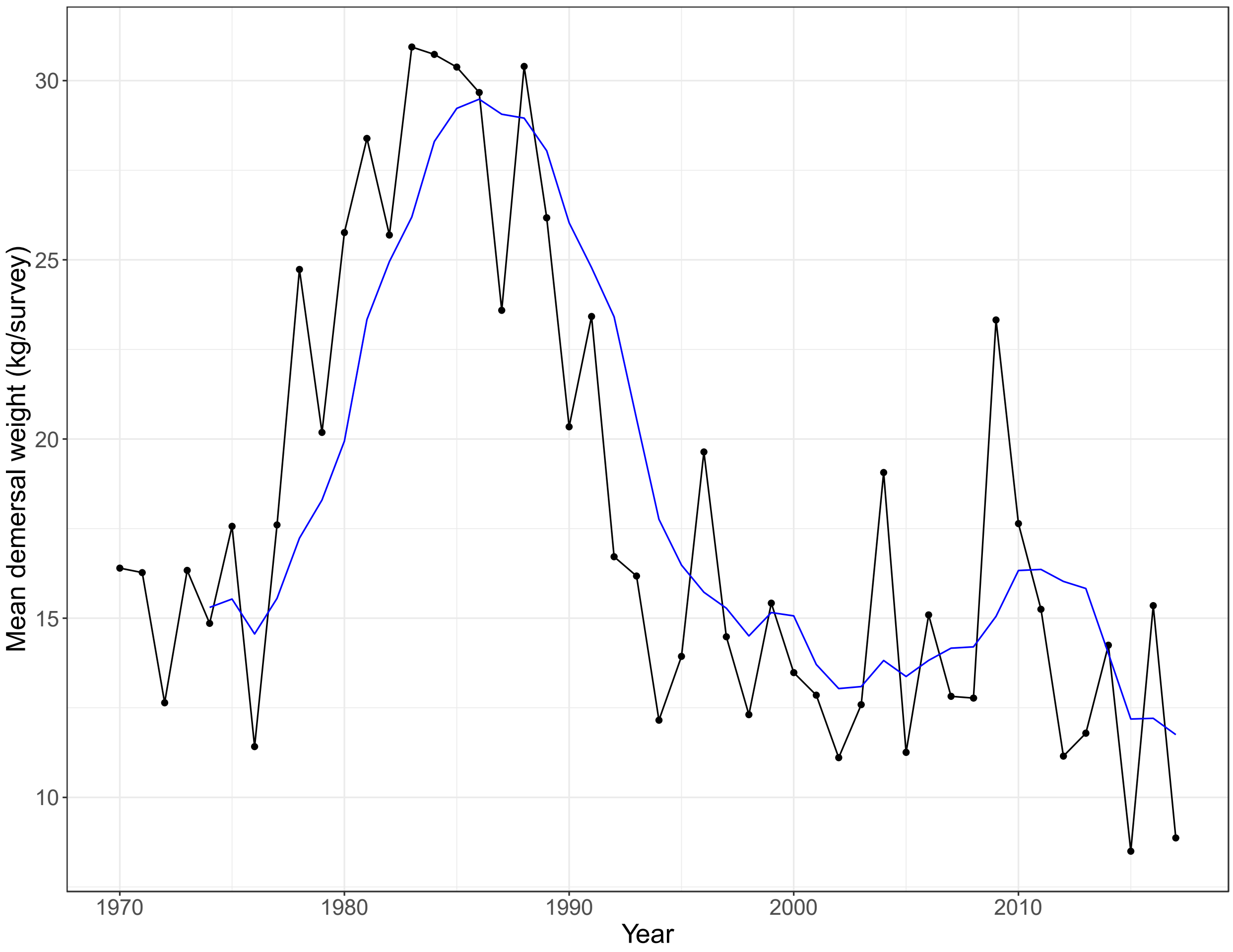

Figure 3.12: Temporal trend of mean demersal weight (kg/survey) on the Eastern Scotian Shelf from 1970 to 2017. Black line represents raw data; blue line represents the five-year moving average.

Demersal:pelagic ratio changes

By weight (kg/survey), the mean ratio of demersals to pelagics declined by 74.4%, leading to assessment as "Endangered" (Figure 3.13; Figure 3.15). The mean ratio of demersals to pelagics peaked in 1974 and declined to its lowest values in 2001 (Figure 3.13). There was an increase in the ratio from 2002 to 2009; however, since 2009 the ratio has been in steady decline (Figure 3.13). As with zooplankton abundance, this indicator is approaching the 80% decline threshold for being categorized as "Critically Endangered" (Figure 3.15).

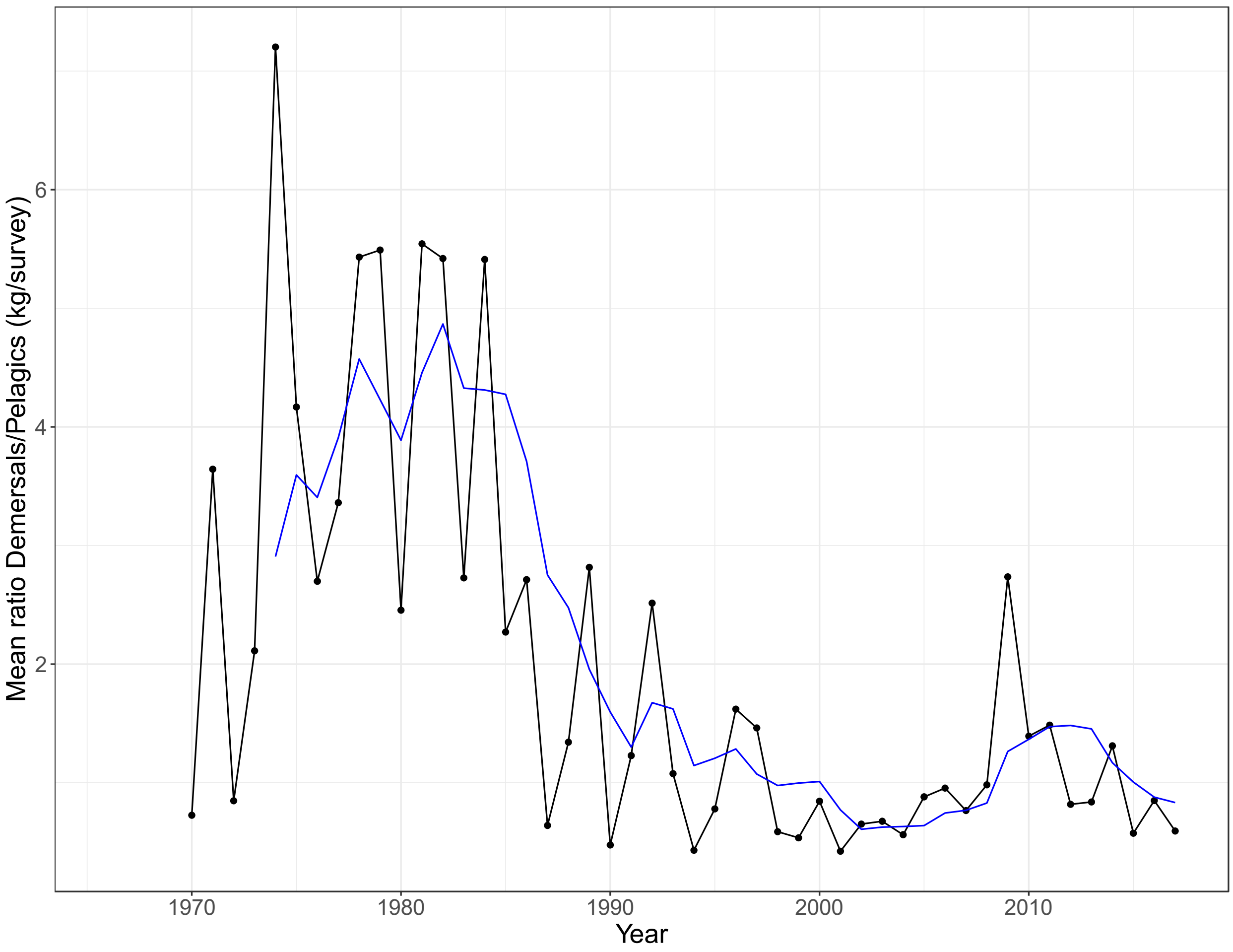

Figure 3.13: Temporal trend of mean demersal to pelagic weight (kg/survey) on the Eastern Scotian Shelf from 1970 to 2017. Black line represents the raw data; blue line represents the five-year moving average.

Grey seals

The five-year mean modelled abundance of grey seals on the ESS increased dramatically by 5,066.8% from 1970 to 2014 (Figure 3.14). As such, under the RLE assessment protocol, grey seal abundance is considered of "Least Concern" (Figure 3.15).

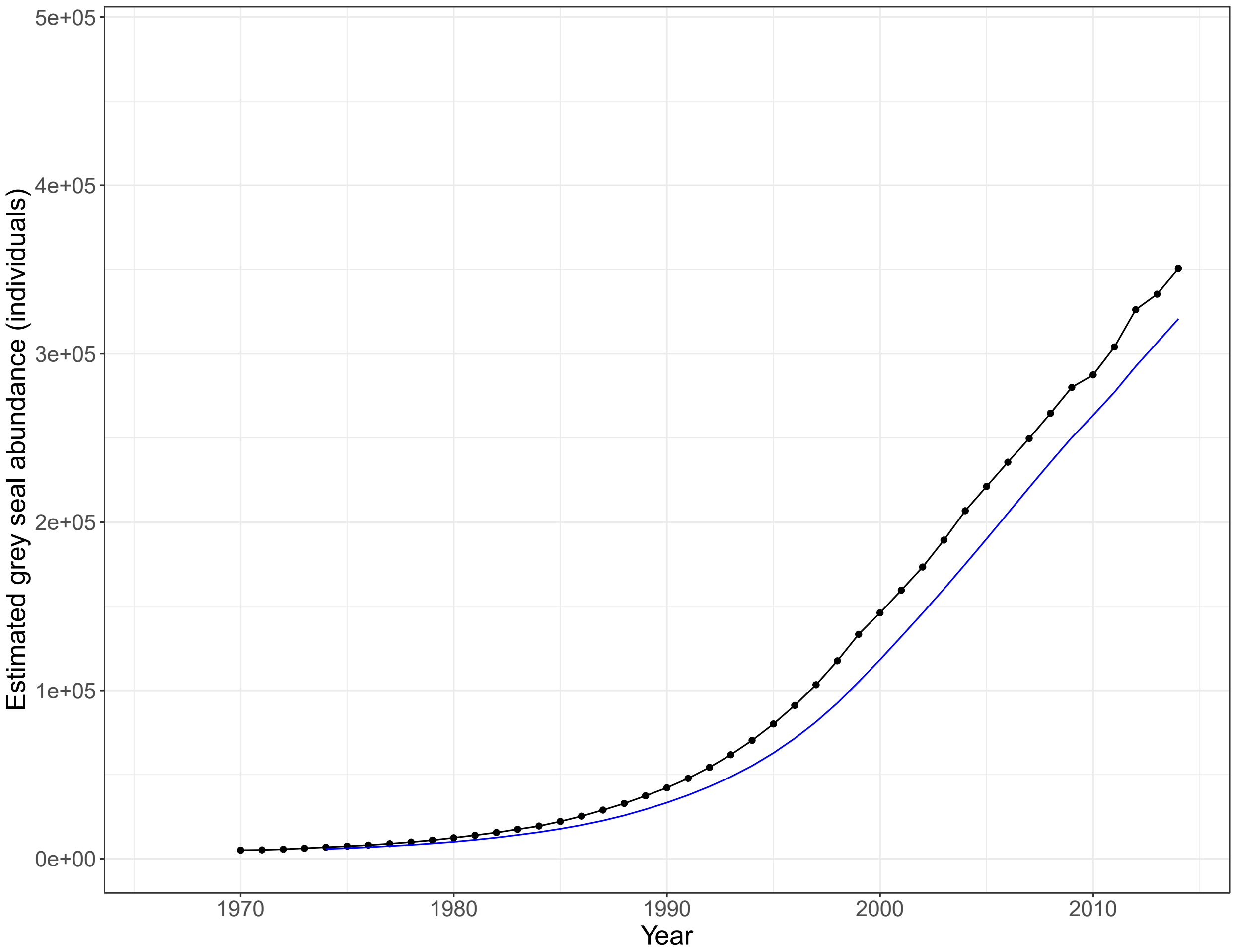

Figure 3.14: Modelled grey seal abundance data for the Eastern Scotian Shelf from 1970 to 2014. Data was provided by Fisheries and Oceans Canada (Hammil et al. 2017) for the years 1970-2014. Black line represents modelled abundance, blue lines represents five-year moving average, grey shading represents 95% confidence interval.

3.3.5 OVERALL RLE ASSESSMENT STATUS

Following the precautionary principle of the IUCN RLE assessment protocol, and based on all of the indicators described above, the decline in biotic components (Criterion D) was categorized as ‘Endangered’ based on both the decline in the ratio of demersal weight to pelagic weight and the decline in large zooplankton abundance between 1970 and 2017 (Figure 3.15, Table 3.5). Given the precautionary principle applied within the RLE, we categorized the current status of the ESS ecosystem as “Endangered” based upon Criteria A-D. However, evaluating the overall risk status of the ESS ideally includes the assessment of Category E, the risk of ecosystem collapse over the next 100 years, which we examine in Chapter 4.

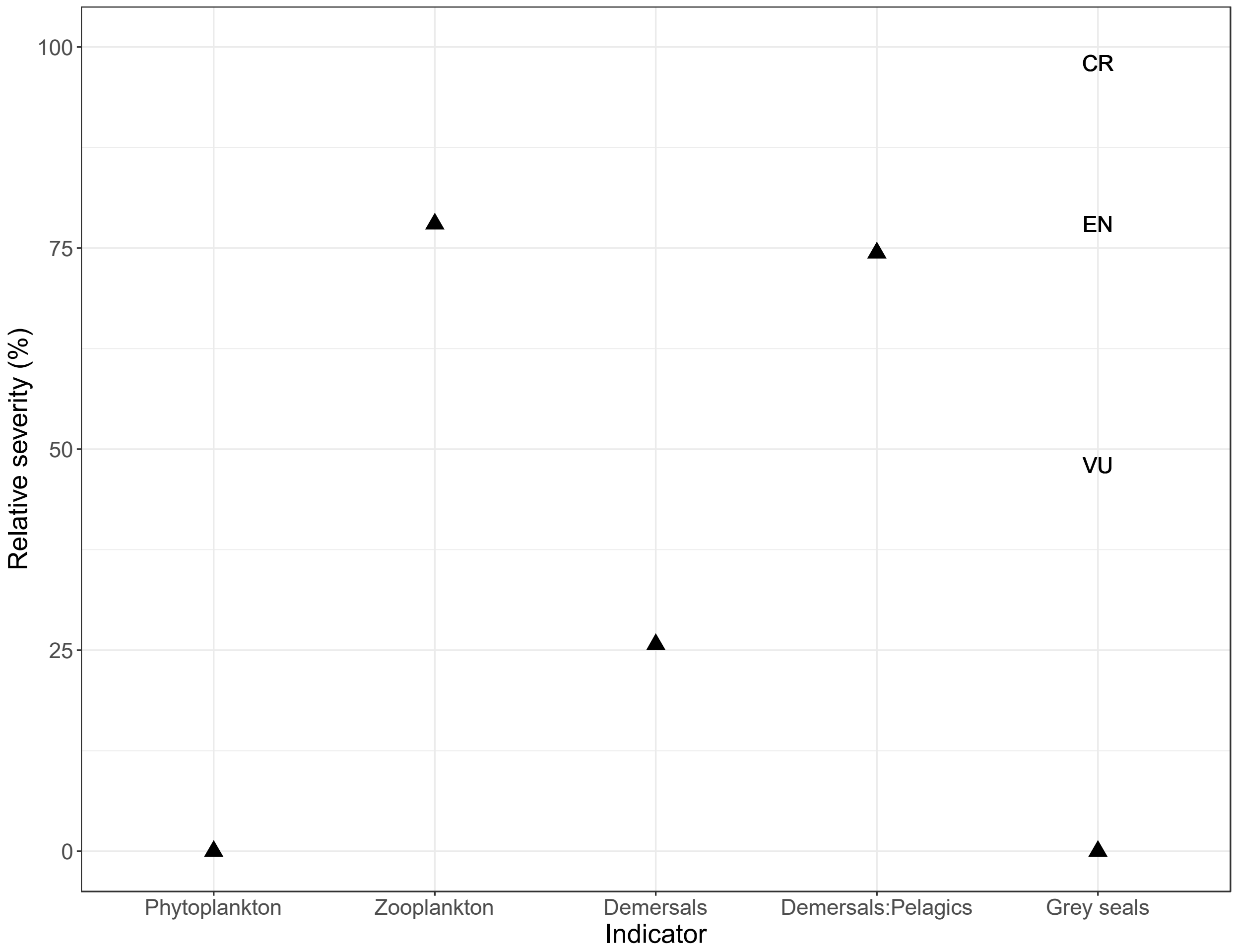

Figure 3.15: Estimates of relative severity for the five biotic indicators of the Eastern Scotian Shelf ecosystem (Criterion D). Triangles represent estimates of relative decline based on collapse thresholds. Increases (e.g. grey seals) were represented by zero. The red box represents the Critically Endangered category (CR); the orange box the Endangered category (EN); and yellow the Vulnerable (VU) category.

Table 3.5: Assessment of the Eastern Scotian Shelf using the IUCN Red List of Ecosystems criteria. DD: Data Deficient; LC: Least Concern; VU: Vulnerable; EN: Endangered; CR: Critically Endangered. Criterion E only has one subcriterion.

Criterion	Declining distribution (A)	Restricted distribution (B)	Environmental degradation (C)	Biotic disruption (D)	Risk of future collapse (E)
Subcriterion 1	VU	LC	LC	EN	See Chapter 4
Subcriterion 2	DD	LC	DD	DD	
Subcriterion 3	DD	LC	DD	DD	

3.4 DISCUSSION

3.4.1 CURRENT RLE RISK STATUS FOR THE ESS

This study applied the Red List of Ecosystems approach to the Eastern Scotian Shelf, an ecosystem that has experienced considerable perturbation during the past 50 years, to determine its risk status and to explore the utility of the RLE for such ecosystems. It expands on the approach used in the only other offshore marine assessment (Bland et al., 2018) by introducing a new method for defining spatial boundaries for offshore marine ecosystems based on their characteristic community compositions and completing the first modelled future projection of an offshore marine ecosystem for the RLE to determine whether assessment Criterion E is feasible in an offshore ecosystem (see Chapter 4). In sum, this has enabled us to develop a reproducible framework for performing RLE assessments for offshore marine ecosystems, which we have provided as an Appendix for use elsewhere (Table A.1).

As the IUCN RLE assessment follows the precautionary principle, the ESS ecosystem was categorized as ‘Endangered’ based on the >50% declines in the ratio of demersals to pelagics and in the abundance of large zooplankton (Figure 3.15; Table 3.5).

We also found that the ESS is experiencing a spatial decline or geographic shift in distribution (Criterion A; Figure 3.7). Given the warming trend in the ESS over recent decades (Bernier et al., 2018), which may favour species with higher optimal temperatures, it seems entirely plausible that the biological community that prefers the relatively warmer temperatures of the WSS (Zwanenburg 2000) has extended its geographic range into the ESS. Alternatively, or additionally, it is possible that the biological community of the ESS is not contracting but rather has shifted northwards to cooler waters in tandem with the expansion of the biological community of the WSS. While our results show that the temperature on the ESS over the past ~50 years has been within the thermal tolerance range of both demersal and pelagic fish in the ESS ecosystem (Figure 3.9), fish can and do shift in response to non-lethal temperature changes, to find more optimal thermal habitat (Pinsky et al., 2020).

The "Endangered" risk categorization indicates the relatively precarious position of the ESS and highlights the potential for further biodiversity loss. However, this categorization is further complicated by the history of the cod collapse on the ESS (Frank et al., 2011). While perhaps an "Endangered" risk status might typically suggest a trajectory towards ecosystem collapse, given the context of the cod-collapse in the early 1990s it is possible that the ecosystem is actually in the process of recovering to its former state and that, in the absence of climate change and under stringent continued fishing restrictions on demersals, this process could continue. However, our study provides no strong evidence of recovery and instead showcases the persistence of ecosystem changes, illustrated by the relatively low levels of demersals and large zooplankton over the latest years of data (Figure 3.11, 3.12). Some studies suggest that the slow recovery of the ESS is in part due to the increasing seal population preying on cod, thus increasing their natural mortality (Sinclair et al., 2015). If this is truly the case, it seems unlikely that recovery can take place even with low fishing mortality given the exponential increase in grey seal numbers. We discuss the ecological and policy implications related to the Endangered ecosystem status of the ESS in Chapter 4.

The interannual, decadal and even multidecadal variability present in marine ecosystems is an additional, or alternative, complication to the ESS risk status. For example, annual recruitment in pelagic species is known to be highly variable (Leggett & Deblois, 1994), and oceanographic variables (e.g. temperature) can show decadal and even multidecadal variability from recurring oceanographic phenomena, such as the Pacific Decadal Oscillation and the North Atlantic Oscillation (Ortuño Crespo et al., 2020). Therefore, a short time-series may misattribute such variability for long term directional changes in an ecosystem. Whilst the time series data used here were nearly 50 years in length - the IUCN recommended timespan for ecosystem assessment - it is still possible that the study captures a cyclical pattern and not a directional change. Therefore, it is important to consider the deep historical context of the ecosystem under assessment.

Unfortunately, but typically for RLE assessments, due to the data demands, our study cannot consider the historical (~1750) ESS ecosystem baseline. Efforts were made to find data to enable a historical assessment to be performed in parallel to the recent past (50 years) assessment. However, the only biotic data available we could find were fishery logs that were used to assess historic cod levels on the Scotian Shelf (Rosenberg et al., 2005). While historical estimates of a single species are insufficient for an IUCN assessment, it can be conservatively estimated that were 1750 the baseline for the assessment, the ESS would be listed as “Critically Endangered” as commercial fishing of cod on the ESS has existed since 1539, and biomass estimates for the Scotian Shelf suggest that cod biomass has declined by >95% since 1852 (Rosenberg et al., 2005). Not surprisingly, the time period chosen influences the final ecosystem assessment categorization and given the known historical human impact on many marine ecosystems (Coll & Lotze, 2016), it is possible that when historical data are unavailable, the RLE categorization is generally an underestimate of the risk status of the ecosystem. Moreover, in RLE assessments that have included historical data (e.g. Gillies et al., 2020; Keith et al., 2013), the historical subcriteria (A3, C3 and D3) always result in higher risk status than the recent past subcriteria (A1, C1 and D1), leading to the ecosystems having a higher overall risk status.

3.4.2 CONSIDERATIONS FOR OFFSHORE MARINE RLE ASSESSMENTS

This study also highlights the particular challenge of assessing spatial declines in offshore marine ecosystems using the IUCN RLE protocol. As noted above, spatial decline can be difficult to quantify for offshore marine ecosystems, as characteristic vegetation or other biotic structures are lacking (at least in the sense that they can easily be detected by, e.g. satellite). Instead, we defined the ecosystem boundary by its biological community. Our study showed that the fuzzy clustering of biological communities shows promise as a technique that can be applied to estimate ecosystem extent and fulfill the requirements for assessments under criterion A1 (Figure 3.7). However, this technique does require more in-situ effort (detailed species/genera annual samples), than the remote sensing analysis that can be used in the assessment of many terrestrial and coastal ecosystems.

Additionally, it is interesting to consider how collapse thresholds of complete loss from a system (e.g. a zooplankton abundance of zero), as is common in assessments to date (e.g. Keith et al., 2013; Lester and Fairweather, 2013; Poulin et al., 2013), affect outcomes. Under these conservative thresholds, ecosystem degradation can only be indicated by declines in biological community properties – increases do not affect the risk categorization. While declines in the characteristic biota of an ecosystem are perhaps the most intuitive warning of ecosystem collapse or change in state, it is worth considering whether significant increases might also signal ecosystem degradation. For example, the grey seal and pelagic fish populations of the ESS have experienced growth for decades (Frank et al., 2011; Sinclair et al., 2015); however, that change would not be identified as a risk to the ecosystem under the RLE assessment protocol. Whilst an increase in abundance is not necessarily a problem, if the energy entering the system remains roughly similar, and there are no changes in body size or food web connections, then that species will be appropriating a greater proportion of the total energy in the ecosystem, and another species must necessarily have declined or decreased in population size, body size, fecundity, or another property. If that impacted species plays an important functional role in the ecosystem, then it could be that the increasing species does indeed signal a transformation of the ecosystem, even if declines are not detected (or monitored). The RLE protocol is not able to directly account for this *per se*. However, as shown here (Figure 3.13) and in Bland *et al.* (2018), it is possible to assess whether increases are, in fact, signals of a trophic restructuring through examining the abundance ratios of characteristic biotic or functional groups. We propose all future IUCN RLE assessments consider incorporating changes in the ratios of functional groups as potential indicators of structural change for Criterion D, as we did in this study with demersals and pelagics. Additionally, we propose future assessments, where the necessary supporting data are available, set collapse thresholds for indicators at above zero. This would allow for increases, in addition to declines to indicate risk of ecosystem collapse and perhaps provide more realistic thresholds of collapse, which likely occur before complete loss of an indicator within an ecosystem.

3.4.3 DEFINING ECOSYSTEM COLLAPSE

It is interesting to note that the RLE assessment protocol defines ecosystem collapse as a transformation into a novel ecosystem with new characteristic species and organizational structure (Bland et al., 2016). This contrasts with other measures of biodiversity loss at the species level, where collapse equates to extinction, but ecosystem collapse does not require the ecosystem to disappear. However, this more conservative definition set out by the RLE assessment protocol (Bland et al., 2016) has rarely been employed: Of the 68 RLE assessments performed, to our knowledge, only the largely dried up Aral Sea has been categorized as "Collapsed" (Bland et al., 2019).

While the ESS has experienced significant changes following the cod-fishery collapse, including dramatic changes to the relative abundance of some functional groups (Figure 3.13), it was only classified as "Endangered." If ecosystems like the ESS, which have experienced restructured relationships between characteristic functional groups do not fit the criteria of collapse, as previously discussed, perhaps it is inappropriate to require a 100% decline of indicators to achieve the risk categorization of "Collapse" as has been common in assessments to date (e.g., Bland et al., 2018; Keith et al., 2013;

Sievers et al., 2020). Ecosystem categorizations might be improved if the IUCN clarified the threshold for collapse.

Finally, it is worth considering whether the IUCN's definition of ecosystem collapse as a transformation into a novel ecosystem with a new organizational structure and characteristic species truly represents ecosystem collapse. As demonstrated by the ESS in Bundy et al. (2004), ecosystem functioning may still be high amidst changes to the ecosystem's organizational structure. Furthermore, as a consequence of climate change, many species are expected to shift their distributions (Pinsky et al., 2020). These climate driven distribution shifts may result in many ecosystems containing new characteristic species while still potentially maintaining ecosystem functioning.

3.4.3 FUTUTRE OPPORTUNITIES

The reproducible framework developed in this study for performing RLE assessments for offshore marine ecosystems (Table A.1) could be applied to other offshore marine ecosystems, and in particular most easily be reproduced in other data-rich offshore marine ecosystems. A good example of this is those other ecosystems frequently surveyed by DFO in Canada (e.g. the WSS). Given the relatively high level of sampling, it may be feasible to categorize many of the offshore ecosystems in Canada's EEZ, which would provide another comparative tool of relative status and risk. In addition, it may also be good practice to perform a RLE assessment of the ESS at frequent intervals, such as every five years, to ensure that any negative trends and opportunities for corrective management are identified early, especially given how close category D indicators are to being categorized as "Critically Endangered" (Figure 3.15) and given the ongoing threat of climate change. While individual indicators can be and are used for assessing ecosystems, the advantages of the RLE, including a consistent approach and categorization scheme that is applied globally, a multi-dimensional assessment, and a format that is easily communicated and grasped by a general audience make it a suitable complement to existing approaches.

3.4.4 CONCLUSIONS

The RLE assessment framework, while necessarily having limitations due to its wide application, is data driven, operates on manageable scales, and can be employed not only in terrestrial and coastal ecosystems, but also in offshore marine ecosystems. Such habitats are perhaps the least intuitive ecosystem type to apply the RLE assessment to, as ecosystem borders are not clearly delineated, so our ability to assess the ESS speaks to the protocol's flexible structure. However, unlike other ecosystem types where remote-sensing data can be used, offshore marine ecosystems require decades of in-situ data monitoring in order to assess spatial decline. The requirement for data from at least two time points over a fifty-year timespan may restrict the application of the RLE framework in data-poor regions, such as the High Seas, and highlights the importance of institutions such as DFO maintaining consistent long-term surveys and data. Despite the challenges in assessing offshore marine ecosystems, the IUCN RLE protocol provides a globally standardized framework that highlights the importance of whole ecosystems as a complement to the RLTS and its species focus. If the IUCN is successful in its stated goal of categorizing all ecosystems on Earth by 2025, it could prove a substantial advance for providing consistent information on relative risk categorization for management, conservation, and policy bodies, and the general public at large.

CHAPTER 4

PROJECTING FUTURE ECOLOGICAL RISK FOR THE EASTERN SCOTIAN SHELF ECOSYSTEM

4.1 INTRODUCTION

4.1.1 FUTURE PROJECTIONS UNDER THE RED LIST OF ECOSYSTEMS (RLE)

The objective of the International Union for the Conservation of Nature (IUCN) RLE is preventing further biodiversity loss and, ultimately, ecosystem collapse by guiding management bodies with a standardized risk assessment and status for ecosystems (Bland et al., 2016). While the immediate ecosystem status is informative in isolation (Criteria A1-D1 and A3-D3; see Chapter 3), the RLE's inclusion of a criterion to consider the trajectory of ecosystems over the 21st century (Criterion E) can provide insight into the potential of future risk to an ecosystem. Incorporating projections of future risk can help decision-makers identify suitable paths forward to help minimize the risk of ecosystem collapse, informed by local or regional trends under relevant scenarios. As with the rest of the assessment process, Criterion E evaluates risk at the ecosystem level, not to individual species (e.g. extinction risk).

To be listed as at risk of collapse under criterion E, the projections must find the probability of collapse to be either (i) ≥ 50% within 50 years (Critically Endangered), (ii) ≥ 20% within 50 years (Endangered) or (iii) ≥ 10% within 100 years (Vulnerable) (Bland et al., 2016; Table 4.1). Unlike criteria A, C and D, criterion E cannot be categorized as "Collapsed," as it is used to evaluate future projections under uncertain scenarios. Regrettably, however, this criterion is rarely evaluated due to the data demands and/or challenges of developing or modifying a stochastic ecosystem model capable of producing probabilistic projections. In particular, while single species estimates of extinction risk (e.g. IUCN Red List) are often available, they are not suitable for the long-term, ecosystem-level assessment needed under Criterion E, which needs to account for interactions among species under modelled scenarios that include societal drivers such as greenhouse gas emission trajectories or concentrations.

While some terrestrial and a few coastal ecosystems have been modelled and their RLE status projected (e.g. Lester et al., 2013), no RLE assessment of an offshore marine ecosystem has yet included an assessment of criterion E. For a protocol that aims to be scalable across ecosystem types, this may represent a weakness, or at least a challenge, if offshore marine ecosystems are particularly difficult to evaluate and project under this criterion. To explore this, we developed a deterministic ecosystem model to project the status of the Eastern Scotian Shelf (ESS) ecosystem to 2100. The ESS is, comparatively speaking, a relatively data-rich offshore marine ecosystem due to long-term monitoring (DFO, 2013), and hence offers a good opportunity to evaluate the broader feasibility of this process by developing a deterministic model, which, if successful, could be modified to include stochastic events and probabilistic projections.

Table 4.1: Thresholds for categorization of Criterion E under the IUCN RLE assessment protocol. Adapted from *Bland et al.* (2016).

Criterion E: Quantitative analysis of the probability of ecosystem collapse	
CR	≥ 50% within 50 years
EN	≥ 20% within 50 years
VU	≥ 10% within 100 years

4.1.2 FUTURE THREATS TO THE EASTERN SCOTIAN SHELF (ESS) ECOSYSTEM

Overfishing and climate change were identified in Chapter 3 as the greatest threats to the ESS. However, subsequent to the Atlantic cod collapse in the 1990s, Fisheries and Oceans Canada (DFO) has maintained strict fishing regulations (bycatch only for many demersal species) on the ESS (DFO, 2015; Sinclair et al. 2015). If these ordinances continue to be observed and enforced, the potential for a fishery-induced collapse, were the environment dynamically stable, seems limited. Climate change, however, is already altering community structures throughout marine ecosystems (Lotze et al., 2019) and can impact the effectiveness of management regimes. Unlike overfishing, the global nature of climate change cannot be controlled or limited by regional managers; instead, they must adapt and can adopt region-specific management schemes informed by ecosystem projections that consider wider changes in climate (Lotze et al., 2019). Despite the existing and expected impacts of climate change, it is rarely considered in management plans for marine protected areas (MPAs) or fisheries management plans (Soomai, 2017; VanderZwaag et al., 2017; Wilson et al. 2020). This is particularly concerning for the ESS ecosystem, given the locally extreme rates of surface water warming in the North Atlantic — expected to increase nearly three times faster than the global average (Pershing et al., 2015; Saba et al., 2016). Longer-term projections of ecosystem stability on the ESS must, therefore include climate-change impacts and their associated risks. We thus focus our projections on the impacts of climate change on the ESS.

4.1.3 PROJECTING THE FUTURE OF THE ESS

In order to project the future of the ESS ecosystem, a model that could capture multiple impacts from individual-level effects to structural changes in the community and ecosystem was needed. For the present analysis, we used *Mizer* (Blanchard et al., 2014; Scott et al., 2014), a size-spectrum ecological model originally created to explore the impact of differing fishing scenarios on community structure (Blanchard et al., 2014). Since its introduction, however, *Mizer* has been modified to enable forcing the ecological community under climate change scenarios (*therMizer)*, allowing for projections of changing community structure under such conditions (Woodworth-Jefcoats et al., 2019). We further modified this model to adapt it to the ESS.

To follow the precautionary principle in terms of evaluating collapse thresholds, we projected the ESS ecosystem under a high emissions scenario (Representative Concentration Pathway (RCP) 8.5) that assumes a continuous increase in the concentration of atmospheric carbon until 2100, with no climate change mitigation (Riahi et al., 2011). As this is the most extreme of the RCP scenarios in terms of atmospheric carbon concentrations and overall climate change impacts, the other scenarios would, in all likelihood find, an equal or lower risk of ecosystem collapse.

4.1.4 OBJECTIVES OF CHAPTER 4

The goal of this chapter was to project the ESS ecosystem it forward until 2100 under a high emission/low mitigation scenario. Additionally, by using the same projected scenario, we aimed to evaluate subcriteria C2 and D2: environmental degradation and biotic disruption within the next 50 years. Finally, we aimed to evaluate the feasibility of completing criterion E and thus conducting complete RLE assessments for other offshore marine ecosystems and provide guidance on how this may be approached. Our study also represents the first application of *therMizer* and *Mizer* to Atlantic Canadian waters.

4.2 METHODS

4.2.1 MODEL

We used the marine ecosystem model *therMizer*, the first size spectrum model with species-level resolution that includes the impact of ocean temperature on metabolic scope (Woodworth-Jefcoats et al., 2019). *TherMizer* builds on and expands the foundation of the *Mizer* model (Woodworth-Jefcoats et al., 2019).

Mizer is a size spectrum model that has primarily been used to model fish communities. It is implemented in the statistical software R using the "mizer" package (Scott et al., 2014). Mizer is not a spatially explicit model, so it assumes that the ecosystem under consideration is homogenous and not patchy in distribution (Scott et al., 2014) and has a default time step of 0.1 years, though projections are only saved every year (Scott et al., 2014).

In *Mizer* body size acts as the "master trait" to constrain individual-level processes such as metabolism, development, and movement, as well as predation (Blanchard et al., 2017). Reproduction is based on a Beverton-Holt type function (Beverton & Holt, 1957) where reproduction nears its maximum as egg production increases (Scott et al., 2014). Predation between individual fish is size-based and is set by a lognormal selectivity function in terms of a preferred ratio of predator to prey weight (Scott et al., 2014). The default preferred ratio of predator to prey weight is 100 (Scott et al., 2014). As predation is size-based, cannibalism is a feature of the model (Scott et al., 2014). Higher-level components such as community structure, biomass distributions and predation interactions then all emerge from these individual level processes (Blanchard et al., 2014, 2017). The only required species-specific trait is the asymptotic weight (Scott et al., 2014). All other species-specific parameters are initially set to defaults within *Mizer* according to ecological theory (see Scott et al., 2014 for further model details), though they can be modified if the necessary data are available (as here). Interactions between individuals are derived from an interaction matrix with a value ranging between 0 (never overlapping) and 1 (identical distributions) for each pair of species. Larvae of all fish species feed on the background resource (plankton community), which is typically calibrated against empirical data where available as the default in *Mizer* is often unrealistic (Scott et al., 2014).

In *therMizer*, species thermal tolerance ranges are added to *Mizer*'s species-specific parameters (Table 4.2), and it is assumed that species become locally extinct outside of their thermal range (Woodworth-Jefcoats et al., 2019). In addition, individual metabolic rate is modified to include a term representing changes with temperature in addition to those with body size (Woodworth-Jefcoats et al., 2019). The effect of temperature on metabolism (TEM) is described by equation (1):

$$\mathrm{TEM}=e^{25.22-E/kT} \qquad (1)$$

derived from the metabolic theory of ecology, where E is the activation energy (0.63 eV), k is Boltmann's constant (8.62×10^{-5} eV K^{-1}), and T is the temperature in Kelvin (Brown et al., 2004; Jennings et al., 2008). In this study, *T* was the projected annual mean sea surface temperature (SST; for pelagics) or bottom temperature (for demersals) derived from an Earth System Model (ESM) (see below). TEM was then scaled (TEM′) for each individual species to be a value between 0 and 1, as in equation (2):

$$\mathrm{TEM}'=(\mathrm{TEM}-\mathrm{Min}_{\mathrm{sp}})/\mathrm{R}_{\mathrm{sp}} \qquad (2)$$

where Min_{sp} is the minimum TEM, and R_{sp} is the range of TEM for each species (Woodworth-Jefcoats et al. 2019). TEM′ then acts as a scaling coefficient applied to the (body-size dependent) standard metabolic rate, causing it to increase with temperature within each species' thermal range (Woodworth-Jefcoats et al. 2019).

In addition to the metabolic impacts of temperature, an additional effect on predation behaviour and encounter rates is also included. Temperatures deviating from preferred values are known to limit effective predation through limiting an individual's aerobic scope (Pörtner & Peck, 2010). To account for this, a temperature effect is added to prey encounter rate (TER) through the following equation (3):

$$\mathrm{TER}=\mathrm{T}(\mathrm{T}-\mathrm{T}_{\mathrm{min}})\,(\mathrm{T}_{\mathrm{max}}-\mathrm{T}) \qquad (3)$$

Where T is the water temperature in Celsius, and T_{min} and T_{max} are the minimum and maximum temperatures of an individual's thermal range, respectively. TER is then scaled to TER′, as described by equation (4):

$$\mathrm{TER}' = \mathrm{TER}/\mathrm{Max}_{\mathrm{sp}} \qquad (4)$$

where Max_{sp} is the maximum value of TER. Therefore TER′, like TEM′, results in a value between 0 and 1 and is used as a scaling coefficient for encounter rate. Practically speaking, in *therMizer* peak prey encounter rate, and therefore predation success occurs at the optimal temperature for aerobic scope (Woodworth-Jefcoats et al. 2019), which here is assumed to be the mid-point of the species' thermal tolerance range. Predation success declines on either side of this optimal temperature to a value of zero beyond the thermal tolerance range (Woodworth-Jefcoats et al. 2019).

TherMizer can also account for the impact of climate change on the plankton community (background resource). Specifically, *therMizer* uses projections of the plankton community from ESMs to represent the impacts of climate change on the plankton community, a critical food resource for fish (Woodworth-Jefcoats et al. 2019). We incorporated projected changes in productivity by replacing the biomass concentration of the original background resource (*kappa*; normally a fixed value over time) with the time series of total plankton community concentrations (phytoplankton plus zooplankton) from the ESM. The magnitude of this time-series was then calibrated against spawning stock biomass through the optimization process (see below).

4.2.2 MODEL PARAMETERS AND CALIBRATION

Following the protocol of Woodworth-Jefcoats *et al.* (2019), we included as many of the 20 most abundant fish species (by empirically observed biomass) as possible based on their data availability, which in this case was observations on the ESS from the annual Fisheries and Oceans Canada (DFO) Research Vessel (RV) survey of the Scotian Shelf from 2017. For the ESS region, nine species of 20 had sufficient data to include, representing around 64.6% of the total observed survey biomass (Figure 4.1). These nine species are listed in Table 4.2, along with their species-specific parameters (e.g. asymptotic weight and thermal range) as derived from the literature (Kaschner et al., 2019; Noble & Clarke 2019a, 2019b). Size at maturity was assumed to be a quarter of the asymptotic weight (Scott et al., 2014). Following the approach of Jabcobsen *et al.* (2017), all values in the interaction matrix were set to 1, meaning all species overlapped in their distribution in the ESS, and therefore predator-prey interactions between species were size-based. While it is true that demersal and pelagic fish often occur in different zones of the water column (Petrik et al., 2019), in order to allow adult pelagics to prey on larval demersals it was necessary to assume overlapped distributions. The remaining model parameters were set to their default values (Scott et al., 2014), with the following exceptions: (i) feeding level, which was increased to 0.8 (from the 0.6 default) in order to ensure all nine species had feeding levels above their critical feeding threshold (the level of food intake that just covers basic metabolic costs, but does not cover reproduction or growth), (ii) species-specific maximum reproduction rate (eggs year^{-1}; *Rmax*), a parameter included in the Beverton Holt reproduction equation (see Scott et al., 2014), (iii) the proportion of energy available for reproduction (*ERepro*), and (iv) the overall biomass of the background (plankton) resource (*kappa*). Following the protocol of Jacobsen *et al.* (2017), we simultaneously calibrated *Rmax*, *Erepro* and *kappa* by minimizing the sum of squared differences between spawning stock biomass estimates for each species from the Ram Legacy Stock Assessment database (Ricard et al., 2012) and catch/landings data for each species from DFO landings records, using a Nelder-Mead algorithm to search the parameter space (R Core Team, 2020).

In *Mizer*, fishing is represented by three parameters: selectivity, catchability, and effort as described in equation (5):

$$\text{Fishing mortality} = \text{selectivity} \times \text{catchability} \times \text{effort} \quad (5)$$

We applied fishing mortality to each species using a “knife-edge” selectivity function where fishing mortality is applied to individuals once they reach their knife-edge size (species-specific size at maturity; Table 4.2). As we were using fishing mortality and not effort, we multiplied this by selectivity and catchability values of one (or zero pre-maturity, given the knife-edge selectivity) in order to impose the observed empirically-based fishing mortality. The majority of observed fishing mortality timeseries were obtained from 1970-2017 were obtained from the Ram Legacy Stock Assessment database (Ricard et al., 2012; see section 4.2.5 for further data discussion).

Multiple hypotheses have been put forward to explain the lack of or limited recovery of Atlantic cod on the ESS following the strict fishing regulations imposed from 1993 onwards, with a consensus that it is driven by an increase in natural mortality, though with disagreement as to the source (Sinclair et al., 2015). Some authors suggest that with the dramatic increase in grey seals (*Halichoerus grypus*) in the region (O’Boyle & Sinclair, 2012; Figure 3.14) there has been a substantial increase in seal predation on juvenile Atlantic cod (Sinclair et al., 2015). Alternative explanations include an increase in forage fish predation on early-life stages of Atlantic cod or an increase in competition for zooplankton between the early-life stages of forage fish and Atlantic cod (Bundy, 2004, 2011). In order to represent the observed increase in natural mortality since the moratorium on directed Atlantic cod fishing in 1993 (Sinclair et al., 2015), in subsequent years we set fishing mortality for Atlantic cod to be 0.5 (Swain & Mohn, 2012) and the knife edge size to be effectively 0g, instead of size at maturity. This captures observed changes without directly specifying the still-debated source of this additional mortality.

Table 4.2: Species-specific model parameters of modelled species. Weights (*w*) are in grams. Maximum weight (W_{max}) was calculated using the length-weight conversions detailed in Appendix Table B.1. Size at maturity (W_{mat}) was set to one quarter W_{max}. The Brody growth coefficient (K_{vb}) was obtained from Boettiger et al. (2012). Maximum recruitment (R_{max}) was optimized following Bland et al. (2014). Minimum temperature (T_{min}) and maximum temperature (T_{max}) were obtained from Kaschner et al. (2019).

Species	W_{max}	W_{mat}	K_{vb}	R_{max}	T_{min}(°C)	T_{max}(°C)
American Plaice (*Hippoglossoides platessoides*)	1683.78	420.96	0.120	1.43E+10	-0.57	19.12
Atlantic cod (*Gadus morhua*)	17573.85	4393.46	0.185	4.28E+08	-0.57	19.38
Haddock (*Melanogrammus aeglefinus*)	3220.89	805.22	0.263	2.51E+09	-0.57	18.8
Atlantic herring (*Clupea harengus*)	282.79	70.70	0.355	9.09E+09	-0.57	25.4
Redfish (*Sebastes sp.*)	3833.47	958.37	0.137	1.27E+10	-0.57	16.64
Spiny dogfish (*Squalus acanthias*)	4499.16	1124.79	0.115	8.86E+09	-0.57	28.51
Turbot, Greenland halibut (*Reinhardtius hippoglossoides*)	4071.22	1017.81	0.078	1.29E+10	-1.22	20.2
Witch flounder (*Glyptocephalus cynoglossus*)	1261.93	315.48	0.217	6.31E+09	-0.24	20.2
Yellowtail flounder (*Pleuronectes ferruginea*)	683.00	170.75	0.335	9.2E+09	1.55	18.98

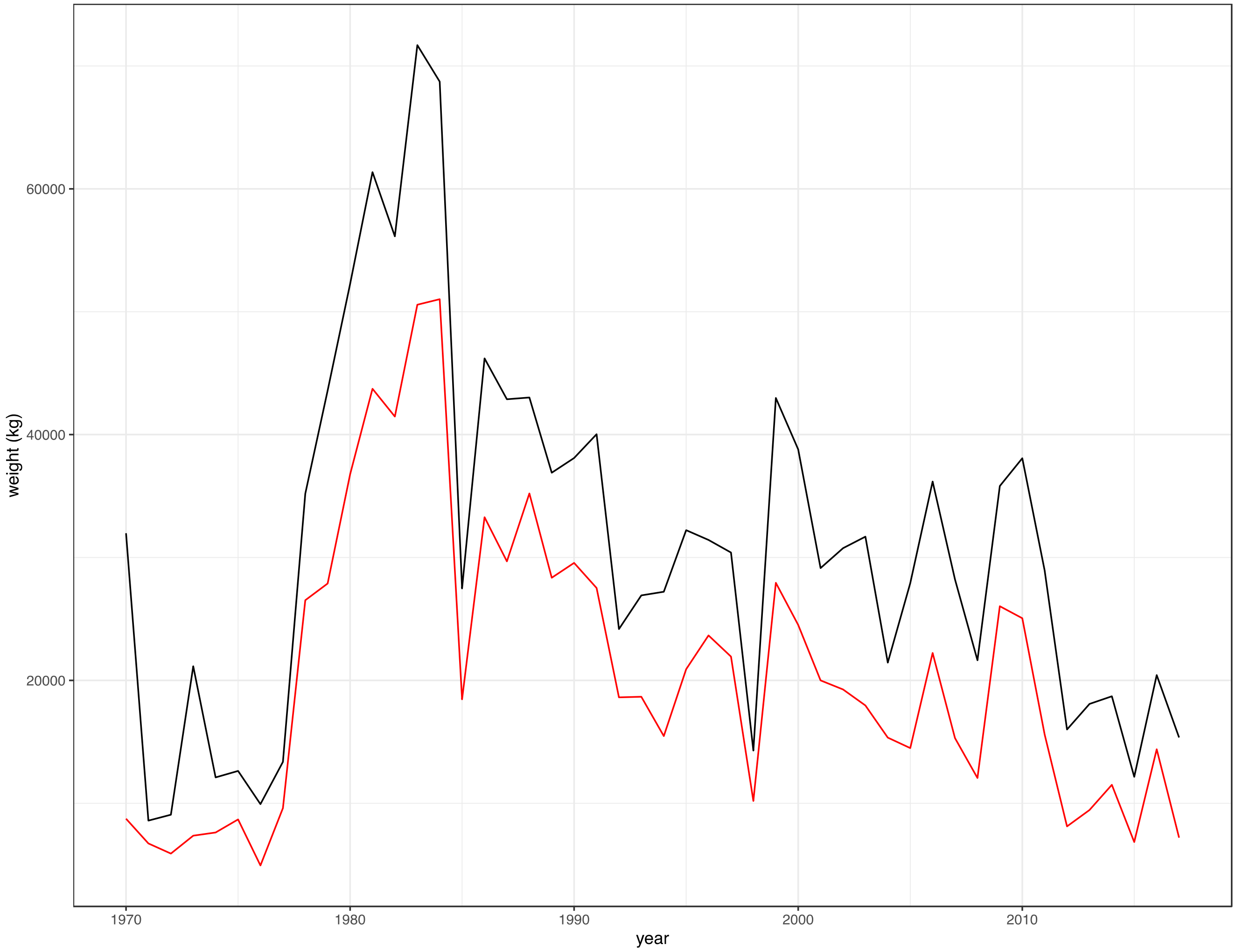

Figure 4.1: Time series of the total biomass (kg) caught in the RV survey (black line) and the total biomass (kg) of the nine modelled species caught in the RV survey (red line) (data provided by DFO).

4.2.3 CLIMATE FORCING VARIABLES

We forced our model with output from the IPSL-CM5A-LR (hereafter IPSL) model, an ESM included in the latest phase of the Coupled Model Intercomparison Project (CMIP6; Eyring et al., 2016), under the greenhouse gas concentration scenario RCP 8.5. CMIP is an internationally coordinated approach for climate and Earth System modelling where multiple models use the same forcing data and scenarios, resulting in an ensemble of climate projections that enable estimates of model uncertainty (Eyring et al., 2016; Woodworth-Jefcoats et al., 2019). Among the many ESMs included in CMIP6, we chose to use IPSL as in previous comparisons it showed relatively strong effects of climate change on oceanographic variables such as sea surface temperature and productivity (Bopp et al., 2013). Likewise, we selected RCP scenario 8.5, a high emissions scenario with no climate change mitigation (Riahi et al., 2011). Together, these make for a worst-case scenario in that if this combination of ESM and RCP scenario does not increase the risk of collapse for the ESS ecosystem in the 21st century, then it is also unlikely that there would be an increase in the risk of collapse with a different ESM and/or under a higher mitigation scenario.

4.2.4 MODEL EVALUATION

The *therMizer* model outputs species-specific biomasses and spawning stock biomasses, as well as catches based on biomass, selectivity, catchability and fishing effort for each species. While spawning stock biomass (SSB) was used to calibrate the model through the optimization process described above (Section 4.2.2), the catch projections can be used to provide an independent evaluation of model performance by comparing them to observed commercial landings data. We compared the model catches from the 1970-2017 period to these data for the same time period (where available) in the ESS ecosystem.

4.2.5 DATA LIMITATIONS

While the ESS is relatively data-rich for an offshore marine ecosystem, with a relatively long time-period of consistent and generally standardized observations, a significant challenge to this study was the quality of data for SSB, catches and fishing mortality. SSB values that were used to calibrate the model are estimates derived from stock assessment models and obtained from the Ram Legacy database (Ricard et al., 2012). Furthermore, for some species SSB estimates for the ESS ecosystem region (Northwest Atlantic Fisheries Organization (NAFO) division 4VW) were unavailable, so the SSB estimate for the nearest region or an aggregate of regions including 4VW was substituted in place (e.g. American plaice SSB estimate is for the NAFO region 4VWX). Additionally, species-specific catch estimates from the model were compared to landings data, as catch data were unavailable from DFO, adding some imprecision to the comparison. When fishing mortality data were unavailable for a species and year, it was calculated using the SSB estimate and landings data using the following equation (6):

$$\text{Fishing mortality} = \text{Landings}/\text{SSB} \qquad (6)$$

Finally, while we required fishing mortality estimates for all species from 1970-2017, some years were missing both fishing mortality and SSB estimates in the Ram Legacy database. In such cases, we held fishing mortality at the value of the previous year. When SSB data or landings data were missing, we left those values empty, as they were used for model calibration and evaluation, not forcing.

These data limitations mean that the study required the use of data that were not always derived from the precise region covered, and/or were modelled estimates with some error associated. Consequently, mismatches between model output and data can be due to issues with the model, the data, or both. Nonetheless, these data represent the best available from our search.

4.2.6 MODELLED SCENARIO

The model was spun-up for 100 years with a static climate, background resource concentration, and fishing mortality (all at the 1970 level), to enable the species biomass estimates to settle. From 1970-2017 the model was forced with temporally varying species-specific fishing mortality data, and IPSL modelled plankton concentrations, mean SST and bottom temperature. We had to force the model with IPSL modelled plankton concentrations as Continuous Plankton Recorder (CPR) data, the primary source of plankton time-series in the ESS, is insufficiently taxonomically resolved for our purposes. Similarly, we chose to use IPSL SST temperature, and bottom temperature as observed temperature records collected by the annual DFO RV survey are only consistently collected for one (warm) month of the year (July; Figure A.1).

The projection period ran from 2018-2100. Fishing pressure was held constant at the average level of 2013-2017. The output from IPSL (SST, bottom temperature and plankton concentration) was used to force the ecosystem model (*therMizer*) (Figure 4.2). *TherMizer* then produced trends of species-specific biomasses, which we assessed over three 5-year averages, 2013-2017, 2046-2050 and 2096-2100 (hereafter referred to as 2017, 2050 and 2100 respectively), to minimize interannual variability.

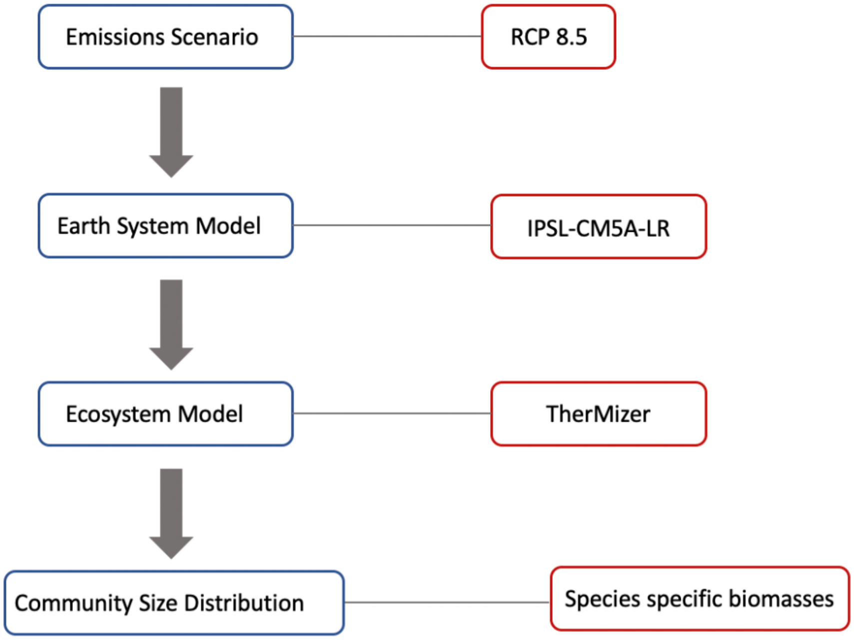

Figure 4.2: Modelling approach used to project the Eastern Scotian Shelf Ecosystem to 2100. The blue boxes describe general approach for projecting ecosystems, and the red boxes our specific method and chosen scenarios/models.

4.2.7 CRITERION E

In order to categorize the future risk of collapse for the ESS ecosystem (Criterion E), it is necessary to evaluate what collapse represents. The IUCN RLE assessment protocol suggests using a probabilistic approach (Table 4.3) to assess this risk within the next 50 and 100 years (Criterion E). However, as previously indicated, criterion E is rarely completed. Given that running multiple probabilistic scenarios and models represents a significant challenge, and the relative paucity of studies even assessing Criterion E, we developed a deterministic model of the ESS ecosystem to determine whether it would be possible to project the future changes to the ESS. If successful, future studies could relatively easily randomly change parameters within our model to add stochasticity.

We focus on using a worst-case scenario to evaluate risk by assessing trends of 5-year averages of species-specific biomasses. While we necessarily then had to modify the thresholds for risk categorization of Criterion E, as the original thresholds are set probabilistically (Table 4.3), we used the same thresholds of risk for categorization as Criteria A, C and D in the RLE assessment (see chapter 3; Table 4.4). Additionally, while Criterion E suggests ecosystem projections should ideally extend for 100 years, we could only project 82 years as the necessary scenarios for our projections were only available until 2100.

Table 4.3: Original thresholds of risk categorization for Criterion E under the RLE assessment protocol. Adapted from *Bland et al.* (2016).

Criterion E: Quantitative analysis of the probability of ecosystem collapse		
	within 50 years	**within 100 years**
CR	≥ 50% risk of collapse	
EN	≥ 20% risk of collapse	
VU		≥ 10% risk of collapse

Table 4.4: Modified thresholds of risk categorization for Criterion E under the RLE assessment protocol. Adapted from *Bland et al.* (2016).

Criterion E: Quantitative analysis of the probability of ecosystem collapse		
	2050	**2100**
CR	≥ 80% decline in all species	≥ 80% decline in all species
EN	≥ 50% decline in all species	≥ 50% decline in all species
VU	≥ 30% decline in all species	≥ 30% decline in all species

4.3 RESULTS

4.3.1 MODEL EVALUATION

Modelled catch and observed landings from 1970-2017, used as a semi-independent evaluation of the model performance, generally correlated well ($r > 0.6$; Table 4.5) for all species except Greenland halibut, yellowtail flounder and Atlantic cod.

Table 4.5: Pearson correlation coefficients for the comparison of modelled and observed landings from 1970 to 2017.

Species	**Pearson's correlation coefficient**
American Plaice (*Hippoglossoides platessoides*)	0.65
Atlantic cod (*Gadus morhua*),	0.48
Haddock (*Melanogrammus aeglefinus*)	0.96
Atlantic herring (*Clupea harengus*)	0.71
Redfish (*Sebastes sp*.)	0.94
Spiny dogfish (*Squalus acanthias*)	0.95
Turbot, Greenland halibut (*Reinhardtius hippoglossoides*)	0.46
Witch flounder (*Glyptocephalus cynoglossus*)	0.73
Yellowtail flounder (*Pleuronectes ferruginea*)	-0.28

While trends in catches were generally well captured, absolute magnitudes of catches often were not (Figure 4.3). While imperfect, *therMizer*, after optimization to the ESS, reproduced the broad trends in fishing observed from 1970-2017. Given that the assessment is based on trends rather than absolute values, we, therefore, determined that it was sufficient to capture broad-scale changes for the ecosystem assessment Criterion E.

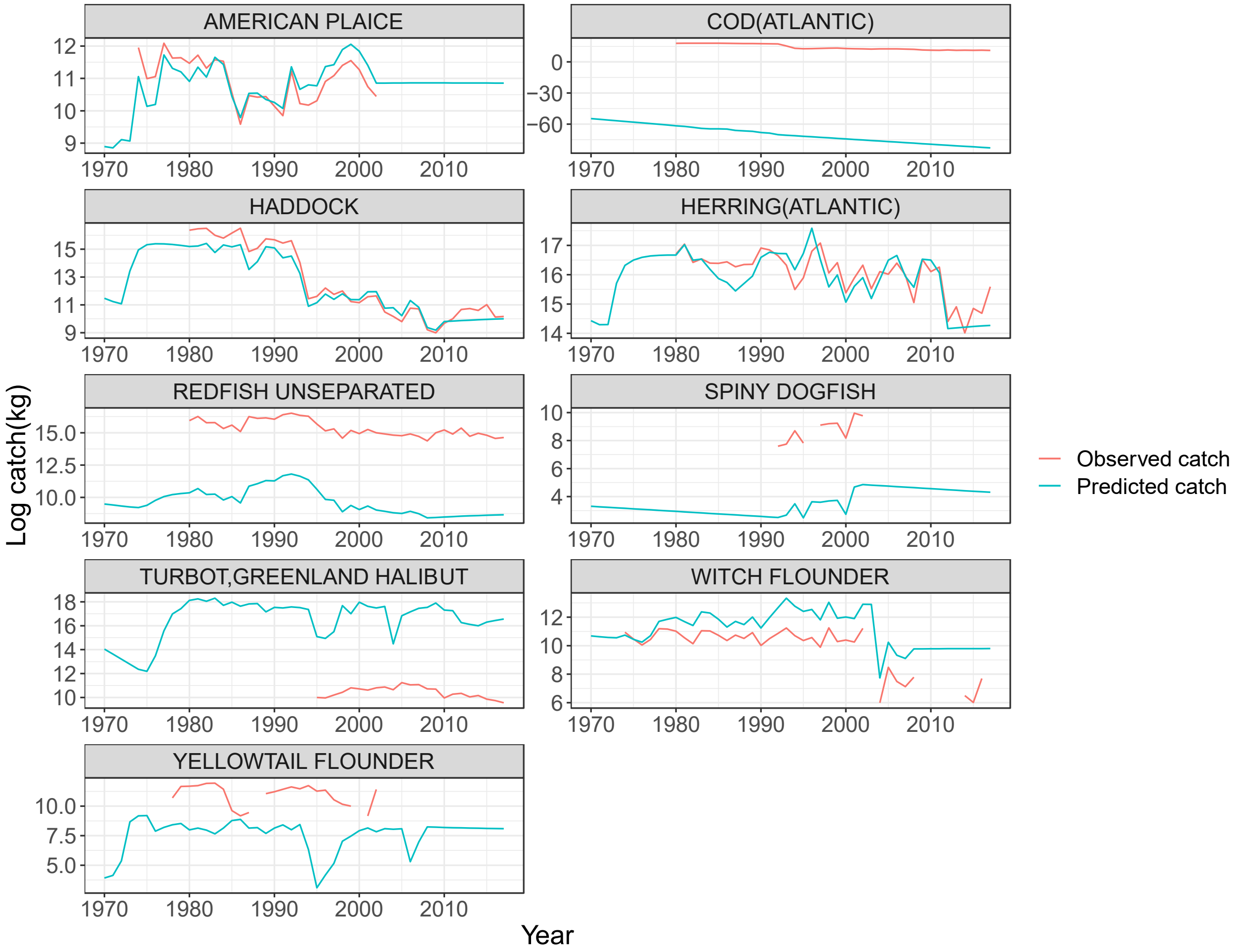

Figure 4.3: Time series of predicted catch and observed landings from 1970-2017.

4.3.2 PROJECTED CHANGES IN BIOMASS

Projected biomasses of individual species in 2050 and 2100 were generally stable relative to 2017 levels, despite climate change impacts (Figure 4.4, 4.5). American plaice, Atlantic cod and witch flounder all experienced changes in biomass of less than 5% by 2100 (Figure 4.4; Table 4.6). The biomass of yellowtail flounder showed only slightly larger changes, with declines of around 6% by 2050 and 8% by 2100 (Figure 4.4; Table 4.6)). Conversely, the projected biomasses of haddock, Atlantic herring, redfish and Greenland halibut increased dramatically by 2100 (~59%, 24%, 72%, and 110%

respectively) (Figure 4.4; Table 4.6). The only species that showed a substantial decline was the spiny dogfish (~74% by 2050 and 95% by 2100) (Figure 4.4; Table 4.6).

Table 4.6: Percent change in species biomasses by 2050 and by 2100.

Species	**Percent change by 2050**	**Percent change by 2100**
American Plaice (*Hippoglossoides platessoides*)	-1.01	-2.85
Atlantic cod (*Gadus morhua*),	-2.46	4.84
Haddock (*Melanogrammus aeglefinus*)	48.13	58.78
Atlantic herring (*Clupea harengus*)	24.66	23.97
Redfish (*Sebastes sp*.)	53.81	72.21
Spiny dogfish (*Squalus acanthias*)	-74.15	-95.19
Turbot, Greenland halibut (*Reinhardtius hippoglossoides*)	107.99	109.91
Witch flounder (*Glyptocephalus cynoglossus*)	3.57	2.75
Yellowtail flounder (*Pleuronectes ferruginea*)	-6.35	-8.15

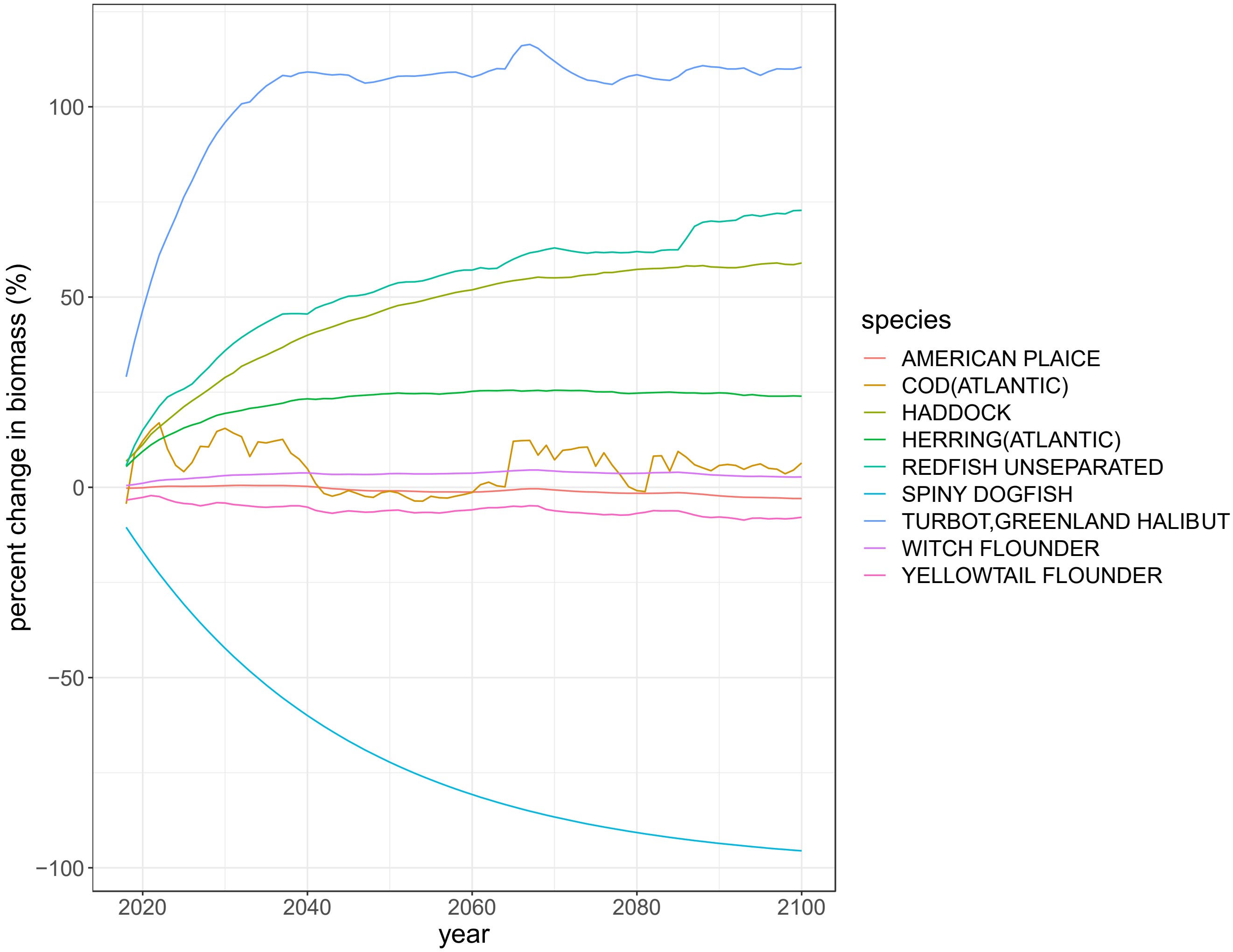

Figure 4.4: Percent change in species biomass, summarized in table 4.6, under a high emissions/low mitigation scenario (RCP 8.5).

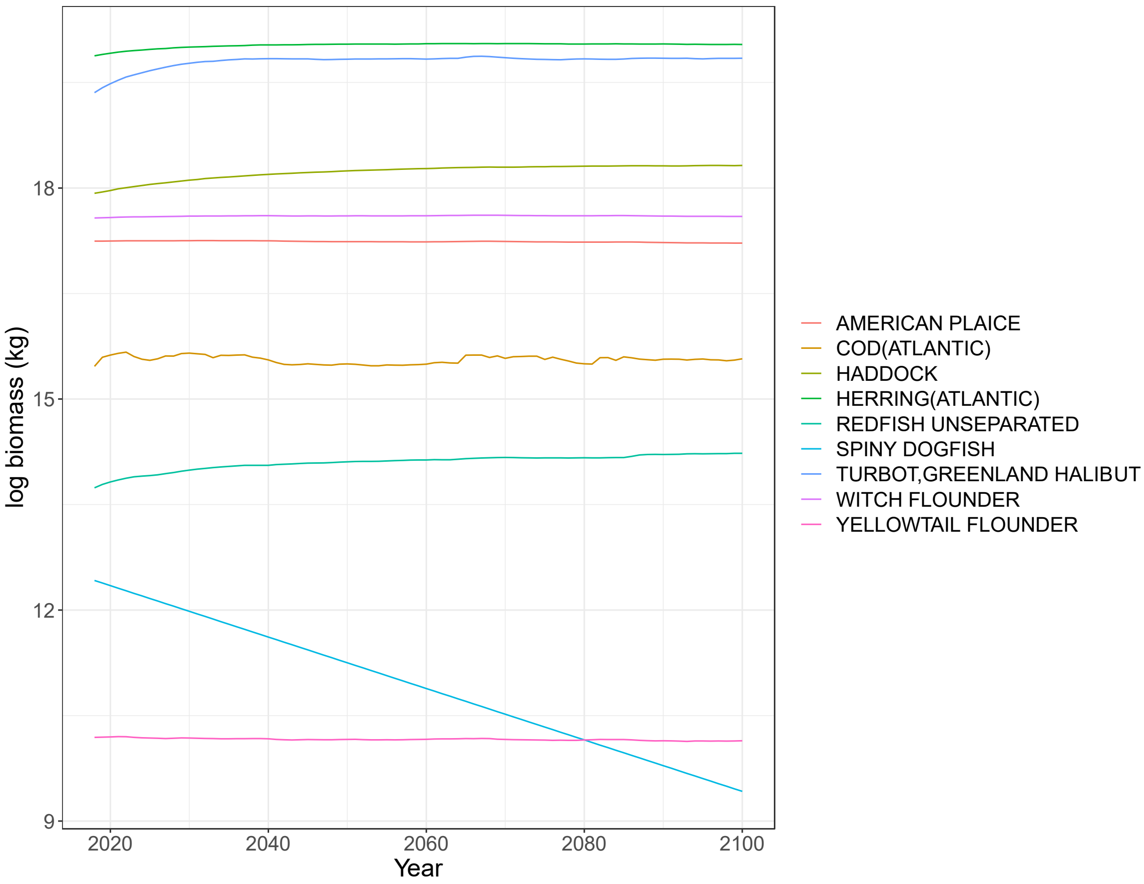

Figure 4.5: Time series of species-specific biomasses from 2018 to 2100 in the ESS, under a high emissions/low mitigation scenario (RCP 8.5).

4.3.3 RLE CATEGORIZATION

Under the high carbon emissions scenario (RCP 8.5), our results did not show mean declines across all species sufficient to meet the categorization thresholds for elevated risk status (Table 4.4) by either 2050 or 2100 (Table 4.6). Therefore, while we did not use a stochastic model, we tentatively categorized criterion E for the ESS as “Least Concern” (Table 4.7). In addition, the ecosystem projection allowed us to evaluate subcriterion C2, environmental degradation, as captured through temperature changes over the next 50 years. As mean projected SST and bottom temperature did not exceed the thermal limits identified in Chapter 3 (Table 3.4) for pelagics or demersals (Figure 4.6, 4.7), subcriterion C2 was also listed as “Least Concern” (Table 4.7). We were not able to project indicators of biotic disruption over the next 50 years due to data deficiencies in species-specific parameters, which limited our projections to nine species. Therefore, D2 was listed as “Data Deficient” (Table 4.7).

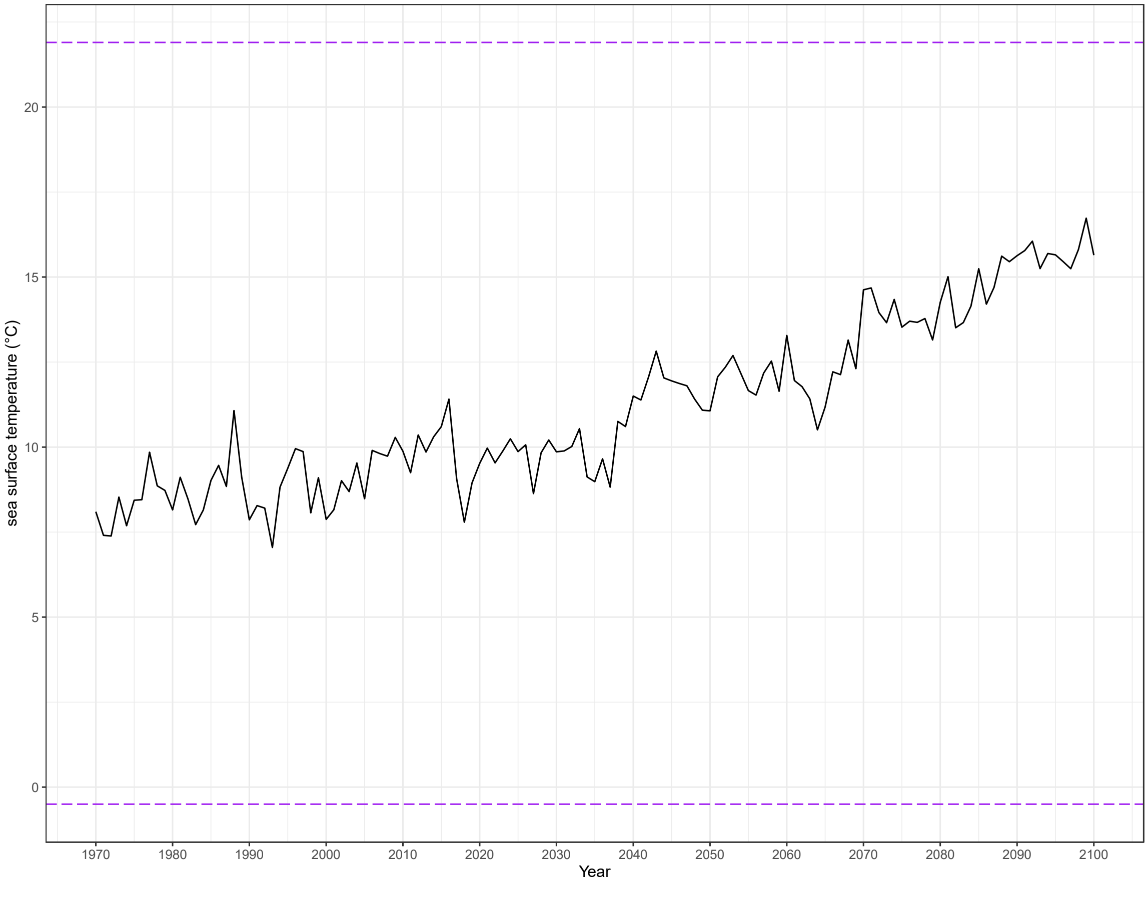

Figure 4.6: IPSL projected mean sea surface temperature for the Eastern Scotian Shelf under RCP 8.5. Purple dashed lines delineate the mean upper and lower thermal limits of pelagics (see Table 3.4).

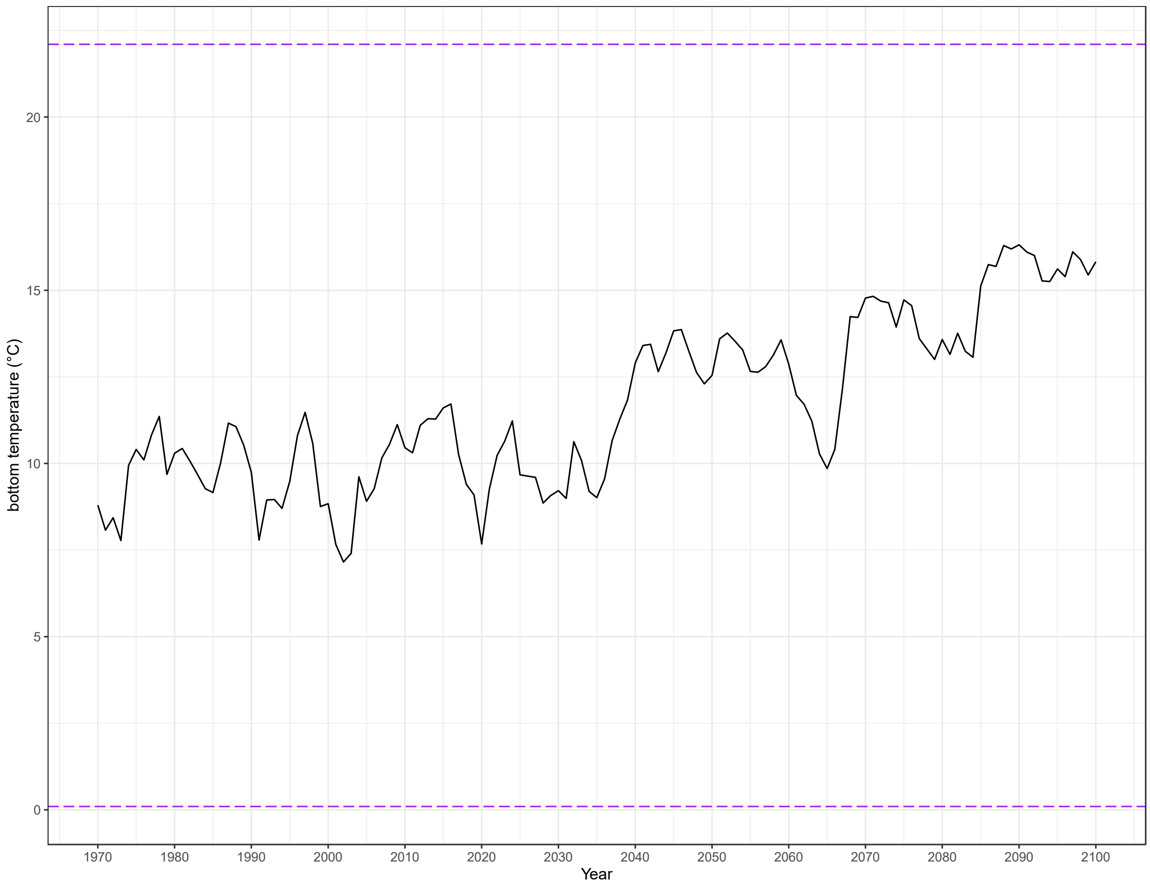

Figure 4.7: IPSL projected mean bottom temperature for the Eastern Scotian Shelf under RCP 8.5. Purple dashed lines delineate the mean upper and lower thermal limits for demersals (see Table 3.4).

Table 4. 7: Assessment of the Eastern Scotian Shelf in 2017 under the IUCN Red List of ecosystems criteria. DD: Data Deficient; LC: Least Concern; VU: Vulnerable; EN: Endangered; CR: Critically Endangered. Criterion E only has one subcriterion.

Criterion	Declining distribution (A)	Restricted distribution (B)	Environmental degradation (C)	Biotic disruption (D)	Risk of future collapse (E)
Subcriterion 1	VU	LC	LC	EN	LC
Subcriterion 2	DD	LC	LC	DD	
Subcriterion 3	DD	LC	DD	DD	

4.4 DISCUSSION

4.4.1 MODEL EVALUATION AND PROJECTION RESULTS

Model evaluation results

In general, the model was able to capture the trends of catch in the ESS from 1970-2017 (Figure 4.3; Table 4.5). The poor correlations of Greenland halibut and yellowtail flounder may be explained by their relatively short time series of observed catch data (Figure 4.3). For Atlantic cod, the poor correlation could be due to imposing fishing mortality on Atlantic cod of 0.5 after 1993 to account for the increase in natural mortality; however, this was necessary to be able to reproduce observed trends in biomass. While no consensus exists on the cause of the increase in natural mortality (Sinclair et al., 2015), without the inclusion of additional mortality, *therMizer* projected a dramatic recovery of Atlantic cod following the decline in fishing mortality observed in 1993.

While the model was able to reproduce the trends in catches, it was not always able to capture the absolute magnitude of catches for some species (Figure 4.3). This may be because, as discussed above (see section 4.2.5), SSB estimates that were used to calibrate the model were sometimes unavailable for the exact area of the ESS (NAFO divisions 4VW) and/or that it is difficult to reproduce observed trends when fishing mortality estimates and observed catches deviate. Nonetheless, as our evaluation of Criterion E was based on relative changes (percentage declines) rather than absolute thresholds, we judged that the model performed sufficiently well to be used for this purpose.

Projection results

We used *therMizer*, a size-structured marine ecosystem model, to assess the potential of climate change to increase the ecosystem risk status of the ESS given projected changes over the coming century. By the end of the projection period (2100), the biomasses of American plaice, yellowtail flounder and spiny dogfish had declined. American plaice and yellowtail flounder experienced reductions in total biomass of around 3% and 8%, respectively. These reductions are relatively small, and hence their total biomasses are projected to remain relatively stable (Figure 4.4, 4.5). The total biomass of spiny dogfish, however, was projected to decline by around 95% by 2100 under the RCP8.5 climate change scenario (Figure 4.4, 4.5). This severe decline is likely due to their relatively high optimal temperature (Figure 4.8), which negatively impacted their ability to successfully forage and compete for the first ~40 years of the projection (Equation 3). We cannot discount possible increases to spiny dogfish biomass if projected beyond 2100, given that the projected mean bottom temperature neared its physiological optimum by 2070, which may explain why its rate of decline slowed. It should be noted that the optimal temperature for spiny dogfish is unusually high for a demersal species. This suggests that the thermal range of spiny dogfish obtained from observational data on Aquamaps may be too wide, and its maximum tolerated temperature (and optimal temperature) is possibly anomalous.

Total biomass increased for the remaining species (Atlantic cod, haddock, herring, redfish, turbot and witch flounder) by 2100. Again, given the constant fishing mortality, this seems to be the result of climate change leading to mean SST and bottom temperatures that approach their thermal optima (Figure 4.8, 4.9), leading to better performance. Importantly, while Atlantic cod biomass increased slightly (~ 4%) during the projection period, it is projected to remain dramatically lower than prior to the 1990's Atlantic cod collapse under this scenario. The positive or ameliorating thermal effects of climate change on the aerobic scope of Atlantic cod, at least from our model projections, will likely be insufficient to lead to recovery of the species in the ESS in isolation. Finally, while the effects of projected decreases in plankton concentration are less obvious in the biomass projections, it is likely that it constrained the biomass changes caused by the warming effects on metabolism and aerobic scope.

While these projections are informative, they are a broad and relatively coarse assessment of the potential impacts of climate change on biomass trends for nine species on the ESS. The resolution is insufficient to capture precisely projected trends, and the timescale long enough to mean that results should be interpreted with appropriate caution and acknowledgement that the uncertainty around projected trends remains unknown. Any single modelling approach is accompanied by uncertainty around the specific implementation of processes and parameter values, and the absence of a projected ecosystem collapse does not guarantee stability— as false negatives are always possible. As such, additional data sources and/or projections should always be incorporated and considered for any decision making or management measures. Options include using multiple scenarios (of both climate change and changing fishing effort), or ensembles of ecosystem models (Tittensor et al., 2018). The data requirements of *therMizer*, as common to most local or regional scale ecosystem models, may restrict its general applicability as a general-purpose model for marine ecosystems. Nevertheless, such models, based on size or functional groups, may be useful if they can identify and warn of possible dramatic ecological changes that are otherwise not visible in species-specific or focussed models.

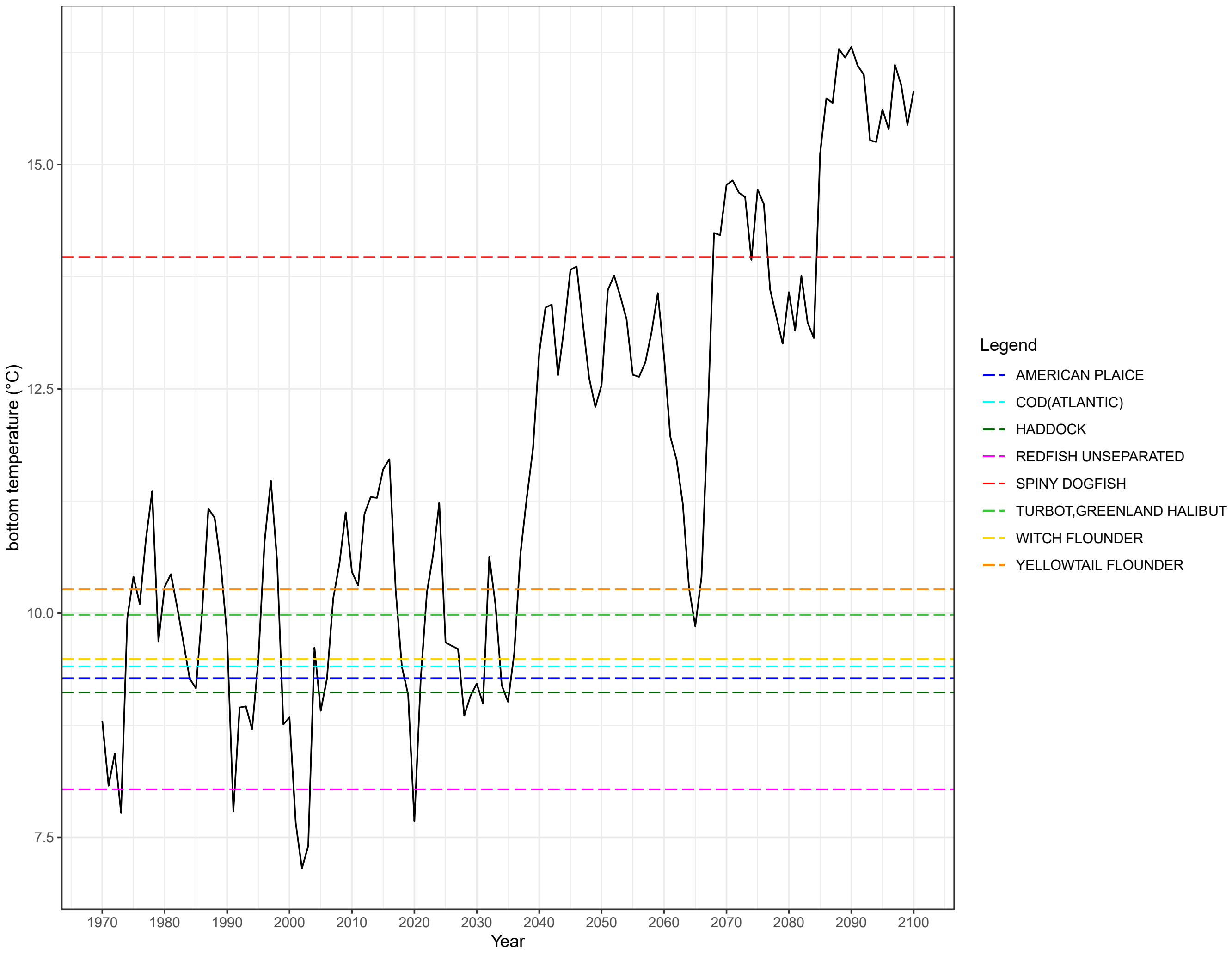

Figure 4.8: IPSL projected mean bottom temperature for the Eastern Scotian Shelf under RCP 8.5. Coloured horizontal lines represent the optimal temperature (mid-point of thermal range) for each species, as defined by *therMizer* (Woodworth-Jefcoats et al., 2019). Thermal limit data was obtained from the Aquamaps observation database (Kaschner et al., 2019).

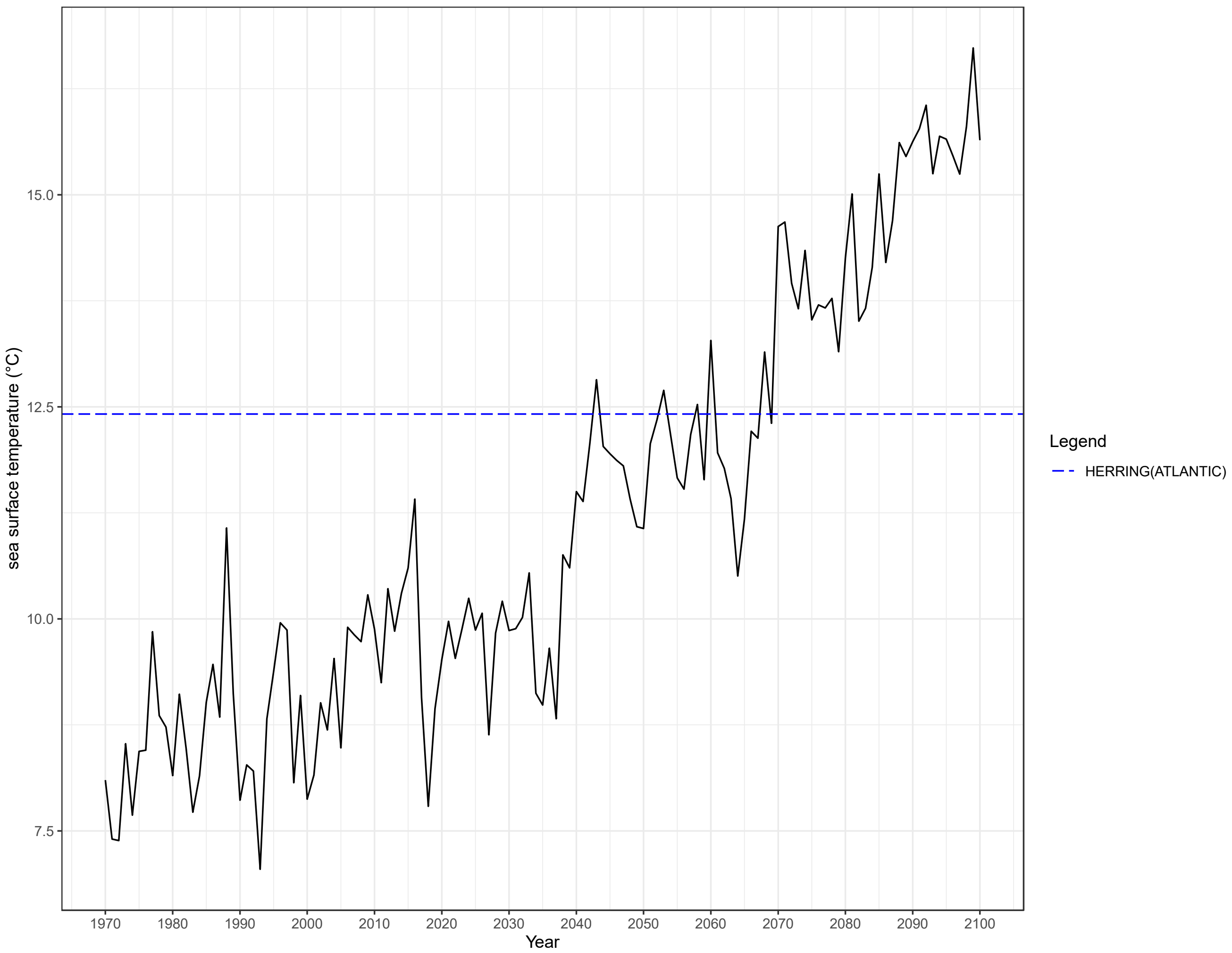

Figure 4. 9: IPSL projected mean sea surface temperature for the Eastern Scotian Shelf under RCP 8.5. The horizontal blue line represents the optimal temperature (mid-point of thermal range) for Atlantic herring, as defined by *therMizer* (Woodworth-Jefcoats et al., 2019). Thermal limit data was obtained from the Aquamaps observation database (Kaschner et al., 2019).

4.4.2 PROJECTION CAVEATS

Data limitations

While an effort was made to build a dataset from multiple sources (DFO and Ram legacy database), we were limited by the data requirements for the *therMizer* model and the species-specific data available for SSB, fishing mortality and catch in the ESS, and therefore were only able to include nine of the top 20 species of fish (by biomass). Including additional species in ecosystem predictions could change the risk categorization, depending on their interactions with existing modelled species, such as predation or competition, and their group-specific thermal thresholds and preferences.

Another data challenge arises from the limited knowledge with which to construct species-specific model parameters for modelled species, including size-at-maturity and growth rate, which influence model projections (Barneche et al., 2019; Blanchard et al., 2014). Underlying much of the uncertainty is the current lack of knowledge of how species life-history characteristics will change in response to distinct climate changes, temperature or otherwise (Woodworth-Jefcoats et al., 2019). Decision-makers would ultimately benefit from more data on species-specific responses being integrated into models derived from the laboratory or in mesocosm experiments or natural observations.

Ecosystem model limitations

Our projection of the ESS ecosystem only included nine species of fish, whereas the real ecosystem actually includes many more non-modelled species of fish and invertebrates, as well as other organisms such as seabirds (Bundy, 2004). In its present form, the size-structured ecosystem model *therMizer* does not easily allow for the inclusion of invertebrates, seabirds, or other endothermic organisms such as marine mammals. This is unfortunate, given the economic importance of invertebrate fisheries such as lobster in the ESS (Frank et al., 2006). This limited our study to a subset of species in higher trophic levels of the ESS ecosystem. In part, this limitation is resolved by calibrating the background resource so as to support reasonable levels of biomass for the species included, but the model will not capture potential interactions with other species that are not included.

Another limitation of *therMizer* is the limited number of climate change impacts captured by the model. Our model projections found that the risk of ecosystem collapse is unlikely to increase given projected thermal increases and changes in the plankton community related to climate change this century for the subset of species that were analyzed and the specific processes and scenario modelled. However, a more comprehensive ecosystem model may consider, for example, the impacts of oxygen concentration, carbon dioxide concentration, and pH on organisms, as well as additional thermal effects such as on disease (Burge et al., 2014; Pörtner, 2012). The ecosystem model used here also doesn't consider how species may redistribute in response to changing environmental conditions (Cheung et al., 2010), which is typical of most size-based ecosystem models.

Scenario limitations

Our projection of the ESS only included one fishing scenario and one RCP scenario. In our projection we kept fishing levels constant at their mean observed levels from 2013-2017. It is unlikely that fishing will remain constant as managers react to changes in biomass of system components. For example, if multiple groundfish species experience increases in biomass as the projections suggest (e.g. Greenland halibut; Figure 4.4, 4.5), it is entirely plausible that fishing regulations, such as total allowable catches, would be loosened. Ideally, it would be good to include a dynamically responsive fishing effort that reacts to changes in underlying biomass, but this makes for a much more complicated dynamically-coupled socio-ecological model with management and economic components, and does not remove the challenge of describing plausible fishing scenarios over the coming century.

While we used the climate change scenario with the highest levels of carbon emissions (RCP 8.5), if climate change mitigation goals are met, and carbon concentration is reduced to levels akin to other RCP scenarios (e.g. RCP 2.6), warming effects on metabolic and aerobic scope will likely not hold and trends may deviate.

Opportunities for future work

As this study has established that it is feasible to project offshore marine ecosystems, future work should exploring stochastic approaches (for example, drawing parameters from distributions) to fulfill the requirements of criterion E. Furthermore, as any individual ecosystem model, such as *therMizer*, includes unique assumptions about how the ecosystem works, future analyses could focus on projecting the ESS using an ensemble of ecosystem models, to better characterize the uncertainty around modelled ecosystem representation (Tittensor et al., 2018). Additionally, this ensemble of ecosystem models could be forced with output from an ensemble of ESMs, themselves forced with a range of RCP scenarios. Implied fishing scenarios could also be applied to the marine ecosystem models. Finally, *therMizer* could be expanded to more easily allow the inclusion of additional species groups and to include other impacts of climate change (see above).

4.4.3 IMPORTANCE OF FUTURE PROJECTIONS (CRITERION E)

No ecosystem assessment framework is flawless, and one criticism of the IUCN RLE assessment framework is that Criterion E often remains incomplete due to insufficient data to construct models (Boitani et al., 2014). Certainly, constructing a model and identifying, sourcing, and integrating data for the ESS here proved time-consuming and challenging. Assessments that have not completed Criterion E often referenced the lack of existing ecosystem-specific projections rather than trying to construct a new model (e.g. Bland et al., 2018; Clark et al., 2015; Keith et al., 2013), suggesting that the workload of constructing a model is a substantial additional burden to the demands of conducting an RLE assessment, especially when it only forms a relatively small proportion of that assessment. A scarcity of data is especially true for marine species, where life history traits and population dynamics are often poorly studied, especially for non-commercial species (Lundquist et al., 2010; Parsons et al., 2014). These data limitations mean that in general RLE risk assessments may be poorly equipped to project risk in response to changes in anticipated threats, such as climate change. But even with limited data, conservation decision making informed by broad ecosystem-specific projections may lead to improved management of marine ecosystems than decision making without such projections. As shown in this chapter, it is possible to project offshore marine ecosystems, at least in a broad sense, when data are limited. And furthermore, models can be revised and improved over time to better capture important ecosystem processes, integrate new data and observations, and contribute to repeated RLE assessments for an ecosystem.

4.4.4 FINAL RLE ASSESSMENT AND RISK STATUS

As we used a deterministic model in this study, we tentatively categorized Criterion E as “Least Concern” as we found that the regional increases in mean SST, and bottom temperature (Figure 4.4, 4.5), and decreases in plankton concentration (Figure B.1) expected under RCP 8.5— high emission/low mitigation scenario— did not perturb the entire system through pervasive metabolic, thermal tolerance, or predation interaction effects, and/or productivity changes, sufficiently enough to reduce the biomass of all species to meet the thresholds to change risk categorization (Figure 4.4, Figure 4.5).

Given this, and following the precautionary principle of the RLE assessment, the final categorization for the ESS, including all of the criteria from Chapters 3 and 4, was found to be “Endangered” (Table 4.7; see Chapter 3 for further discussion of other criteria). As this is the first assessment of an offshore marine ecosystem that completed assessments under all RLE assessment criteria, we have provided a reproducible framework for RLE assessments of offshore marine ecosystems that can be applied elsewhere (Table A.1)

4.4.5 IMPLICATIONS OF RLE ASSESSMENT FINDINGS

As an “Endangered” ecosystem, the ESS is considered to be at very high risk of collapse under the RLE (Keith et al., 2013). This categorization is analogous to the Red List of Species, where an “Endangered” species is at high risk of extinction (IUCN, 2012). Both advise managers to act to reverse the trajectory of biodiversity loss (Bland et al., 2016; IUCN, 2012; Keith et al., 2013). Our study found that ecosystem collapse is most likely to arise by a further decline in the ratio of demersals to pelagics, the declining abundance of zooplankton, or the spatial distribution of the community (Table 4.7; Figure 3.15). These are aspects of the ESS ecosystem that warrant further consideration by managers. Potential management actions to lower the risk status of the ESS could include lowering the bycatch limits of demersal species, namely Atlantic cod and haddock, for commercial fisheries. Additionally, as only a limited number of climate change impacts are captured by the model, resource managers should proactively enact adaptation strategies to dampen any potential effects of climate change, such as reducing other stressors on the ESS ecosystem (Wilson et al., 2020).

The culmination of overfishing in the 1990s drove dramatic restructuring of the ESS ecosystem (Frank et al. 2011; see Chapter 3). As our study has a baseline from before this period, it is possible that the ecosystem is in the process of recovering to its former state, though recent trends suggest otherwise (Bernier et al., 2018). Furthermore, the trends for biotic indicators (Criterion D; see chapter 3) from 1993-2017 do not suggest recovery (Figure 3.10, 3.11, 3.12, 3.13, 3.14). Regardless of the trajectory, the categorization of the ecosystem as “Endangered” represents significant degradation and slow recovery as observed in the decades following the 1990s Atlantic cod collapse (Hutchings & Rangeley, 2011). More management tools may be necessary to enable ecosystem recuperation, such as reducing the abundance of grey seals (O’Boyle & Sinclair 2012; Sinclair et al., 2015).

4.4.6 CONCLUSIONS

It is imperative that risks to biodiversity are assessed beyond the species level (Keith et al., 2013). Evaluating ecosystem-level biodiversity risks can link abiotic and biotic components to identify specific drivers of degradation at scale (Noss, 1996). The RLE is a prominent framework that has been developed to standardize ecosystem-level observations for global comparisons of risk (Bland et al. 2016; Keith et al., 2013). By addressing criterion E in an offshore marine ecosystem, in this chapter we showed that completion of the RLE assessment protocol is possible not only in terrestrial and coastal ecosystems but also offshore. This is crucial as pelagic ecosystems cover over 66% of Earth's surface (Angel, 2003). To aid future assessments of this realm, we have provided a reproducible framework for an RLE assessment for offshore marine ecosystems (Table A.1). Finally, this chapter solidifies the findings discussed in Chapter 3, confirming the classification of the ESS ecosystem as "Endangered" under the RLE, and thus deserving of the attention of resource managers, conservationists, and decision-makers to limit or prevent further loss of biodiversity, and to facilitate its recuperation.

CHAPTER 5

CONCLUSION

5.1 SUMMARY OF THESIS

Biodiversity loss increasingly threatens the stability of the biological functioning responsible for many critical ecosystem services (IPBES et al., 2019a; Mace et al., 2012; Worm et al., 2006). Understanding the strengths and limitations of the various biodiversity policy tools that have been designed in an attempt to measure and combat this loss is critical (Burgass et al., 2020). In this thesis, I assessed the suitability of two preeminent biodiversity policy and management tools, operating at global and regional scales, respectively, for evaluating risk and informing societal responses. In Chapter 2, I examined the uptake of the biosphere integrity planetary boundary into environmental policy in the decade since its inception, using a comprehensive literature review. My results showed that the biosphere integrity planetary boundary framework has been infrequently incorporated, at least overtly, into policy tools (Hurley & Tittensor 2020). Yet, it is unclear whether this uptake rate is typical for environmental science concepts as few studies to date have examined the rate at which such science-driven approaches are incorporated.

Chapter 3 developed an IUCN RLE assessment of the ESS, a prominently perturbed offshore marine ecosystem which underwent a fisheries collapse in the 1990s (Frank et al., 2011). This study is among the first to complete an RLE assessment in an offshore marine ecosystem, as it has almost exclusively been applied to terrestrial and coastal ecosystems. The RLE protocol relies, in part, on determining whether the ecosystem under assessment has experienced a spatial decline, which can be ambiguous in offshore marine ecosystems where conventional monitoring tools like remote sensing images are not necessarily suitable. However, I developed approaches to deal with this challenge, and the availability of biological time series along the ESS afforded the use of specific statistical tools (namely fuzzy clustering) to probabilistically identify the spatial contraction of indicator assemblages. Furthermore, I found that the ESS ecosystem appears not to have returned to its pre-fisheries collapse state, based on the most recent data that I was able to assess.

In Chapter 4, to complete the RLE protocol, I developed a regional model built on *therMizer*, a size-structured ecosystem model able to incorporate climate change impacts on temperature and productivity. I then used this model to project the state of the ESS ecosystem to 2100 under the IPCC “business as usual” carbon emissions scenario to determine if this would affect its risk category under the IUCN RLE. These projections showed that expected increases to mean SST and bottom temperature under a business as usual carbon emissions scenario did not exceed the thermal thresholds of demersals or pelagics, nor perturb the system through metabolic or interaction effects in such a way as to increase the likelihood of a collapse.

Ultimately the results of this RLE assessment (chapter 3 and 4) resulted in a categorization of the ESS ecosystem as “Endangered,” implying the ecosystem is at high risk of collapse, and managers should act to reverse the trajectory of biodiversity loss. To accompany my risk assessment of the ESS, I have created a reproducible framework for performing RLE assessments (Table A.1) in other offshore marine ecosystems, based on my experiences and the approaches that I developed.

5.2 FUTURE RESEARCH

The findings of this thesis reveal several promising future research avenues. Firstly, if the biosphere integrity planetary boundary framework is to be more frequently adopted into policy, it should be downscaled and implemented at national or other regionally manageable scales. Downscaling the biosphere integrity planetary boundary could improve its contributions to the global initiative to slow biodiversity loss. It might also be instructive to apply our standardized search terms to examine the uptake of the other eight planetary boundaries into policy and evaluate whether and where the challenges of policy adoption differ. More broadly, Chapter 2 identified a gap in studies addressing the rapidity with which science is embedded into policy. If this field received more attention, common attributes among the most rapidly embedded science ideas into policy could be identified, and scientists could develop their work using a framework most likely to impact policy.

Beyond the suggestions of future work discussed in chapter 4 (see section 4.4.2), an additional area for further consideration arising from Chapters 3 and 4 is that the IUCN RLE assessment definition of ecosystems, as a physical environment with interacting organisms that can be impacted by humans (e.g. fishing, climate change) (Bland et al., 2016), does not account for the impact of ecosystems on humans and human systems. This missing component leads to a definition of ecosystem collapse that does not include or cannot be predicated upon the associated loss of ecosystem goods and services that humans rely on. While depleted biotic richness and aberrant abiotic conditions are hallmarks of ecosystem collapse, we must also consider the associated impact of diminished socio-economic benefits. Perhaps socio-economic benefits could be added as an additional indicator under existing RLE criterion D (biotic disruption) with a threshold when the ecosystem is no longer able to provide the goods and services human societies rely on. This would mean that collapse or ecosystem status could also be based on concomitant impacts on the human societies that depend on them, which seems relevant both to the need for sustainable utilisation of resources from ecosystems but also to the way that we sometimes designate these reversed impacts in similar ways (e.g. the ‘fisheries’ collapse in Newfoundland; Hutchings & Myers, 1994). Alternatively, the examination of the risk status of socio-economic ecosystem goods and services could be an optional but meaningful supplemental step to the RLE assessment protocol.

Either way, this alternative threshold may have to interpret collapse differently for each region and consider their unique needs from ecosystem services, necessitating a fairly in-depth protocol. This suggests that it could perhaps be an additional or separate assessment, rather than integrated into the existing one. The socio-economic consequences from a significant decline in ecosystem functioning are well known. Infamous among these examples is the Newfoundland cod collapse in which 30,000 fishers and fish plant workers lost their jobs (Higgins, 2008), with the ramifications still felt today. An evolving example is the critical decline in many sockeye salmon runs in British Columbia (CBC, 2019). These declines have been identified not only by scientists, but also by fishers, with more than 2,500 salmon fishers’ incomes greatly reduced (Matti, 2019). For the ESS, a socio-economic signal of the collapse was the decline of cod and a forced halt in fishing, and a shift to targeted species in lower trophic levels (e.g. herring). As such, it would be interesting to consider the addition of a category ‘F’ considering socio-economic impact to the RLE assessment protocol (Figure 5.1).

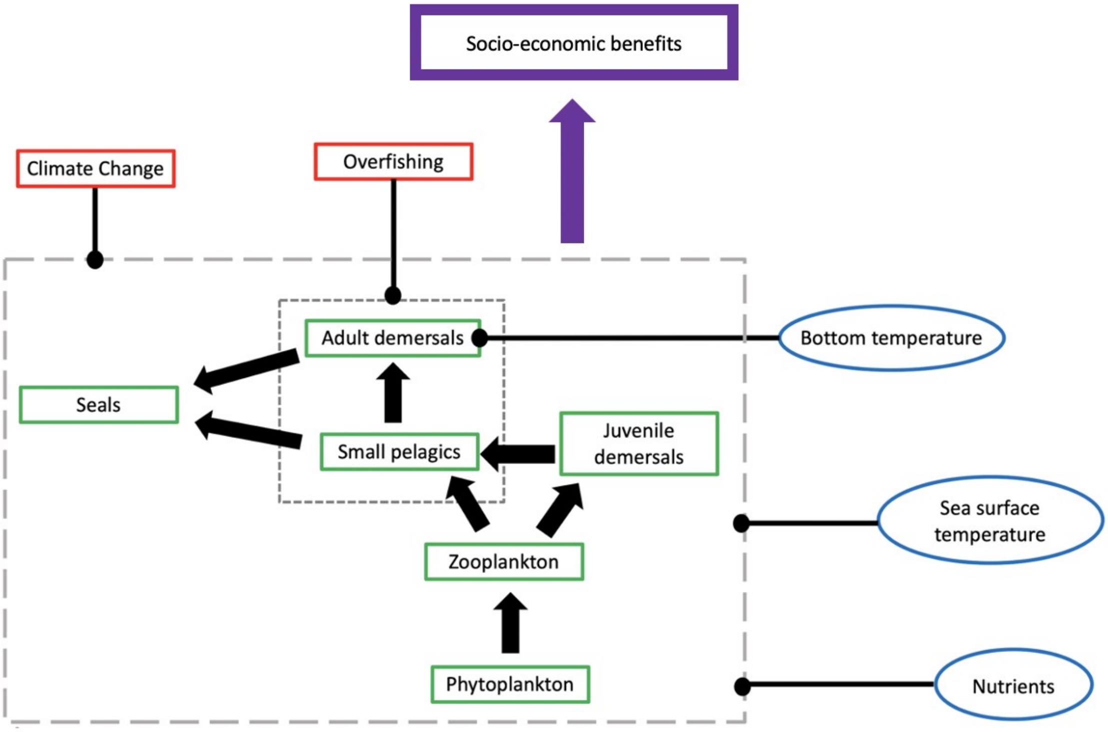

Figure 5.1: Conceptual model of the processes and interactions relevant to the Eastern Scotian Shelf ecosystem risk assessment. Blue ovals indicate environmental processes, green boxes biotic components, red boxes threats and purple boxes socio-economic benefits from the ecosystem. Hatched boxes represent ecosystem components susceptible to threats.

5.3 CONCLUSION

The pace of biodiversity loss is accelerating (IPBES, 2019a), and the risks posed to human society have never been clearer. Mitigation initiatives will rely on robust biodiversity policy and management tools, two of which were examined in this thesis to evaluate their effectiveness. We found that the biodiversity planetary boundary framework has had modest uptake into policy, and in order to increase its effectiveness there should be an effort to increase its policy incorporation beyond Europe. The biodiversity planetary boundary has yet to be quantified, prompting debate about the existence of thresholds or boundaries of biodiversity loss (Montoya et al., 2018a, 2018b). Efforts to quantify the planetary boundary and downscale it to the national level would likely increase the effectiveness of the framework in policy and potentially ease the debate about the existence of such boundaries. We also found that that all criteria within RLE assessment protocol can effectively be applied in not only terrestrial, freshwater and coastal ecosystems, but also in offshore marine ecosystems. The RLE's suitability in this diverse set of ecosystems suggests it could be an effective tool for biodiversity policymakers in the future. In conclusion, integrated frameworks such as the two examined in this thesis, are critical to support informed conservation efforts for maintaining biodiversity, ecosystem stability and to avoid functional boundaries and detrimental impacts on humankind.

APPENDIX A: SUPPLEMENTARY MATERIALS FOR CHAPTER 3

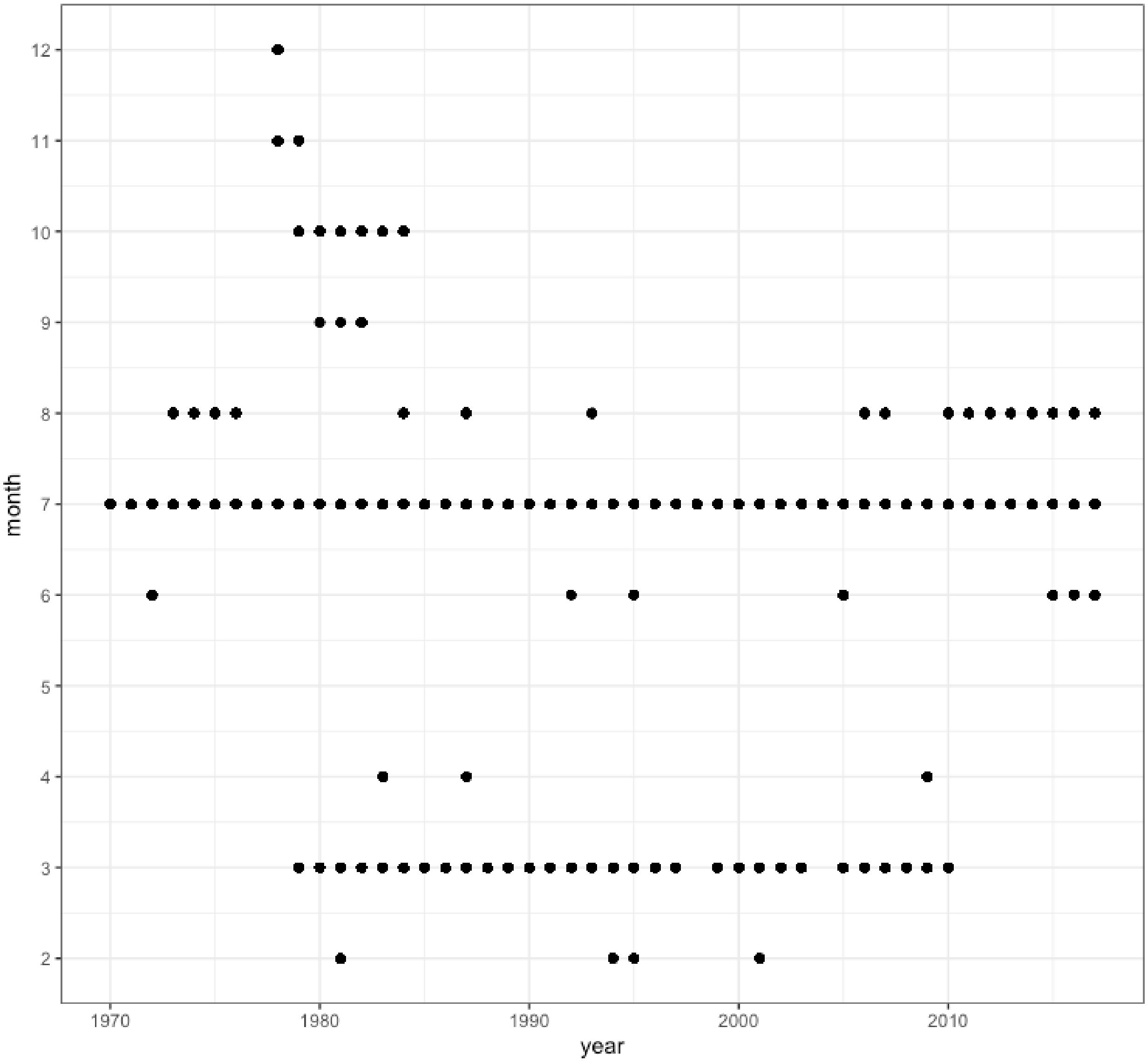

Figure A.1: Months (January = 1, December = 12) of occurrence for the Fisheries and Oceans Canada Research Vessel Survey of the Scotian Shelf from 1970-2017.

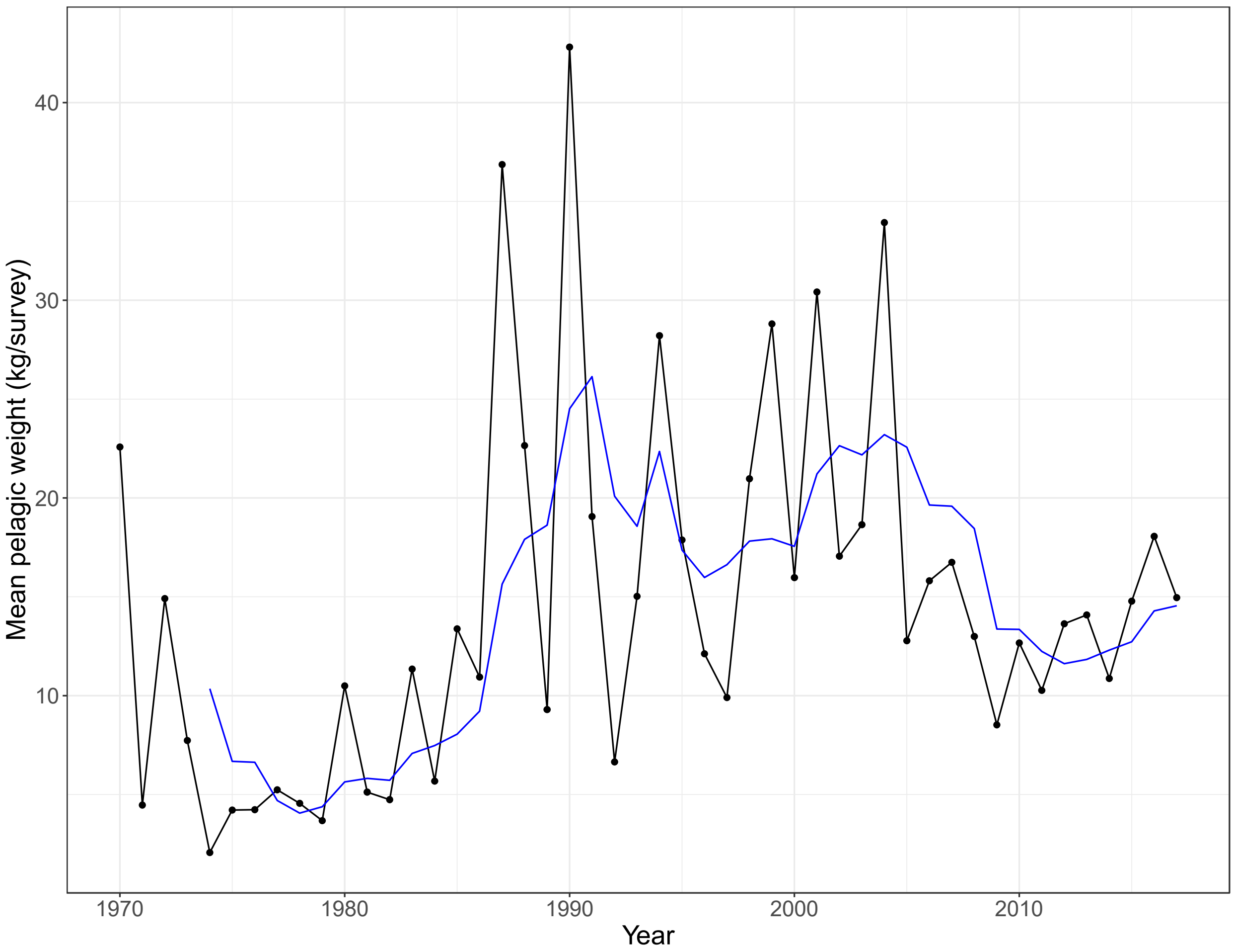

Figure A.2: Temporal trend of mean pelagic weight (kg/survey) on the Eastern Scotian Shelf from 1970 to 2017. Blue line represents the five-year moving average.

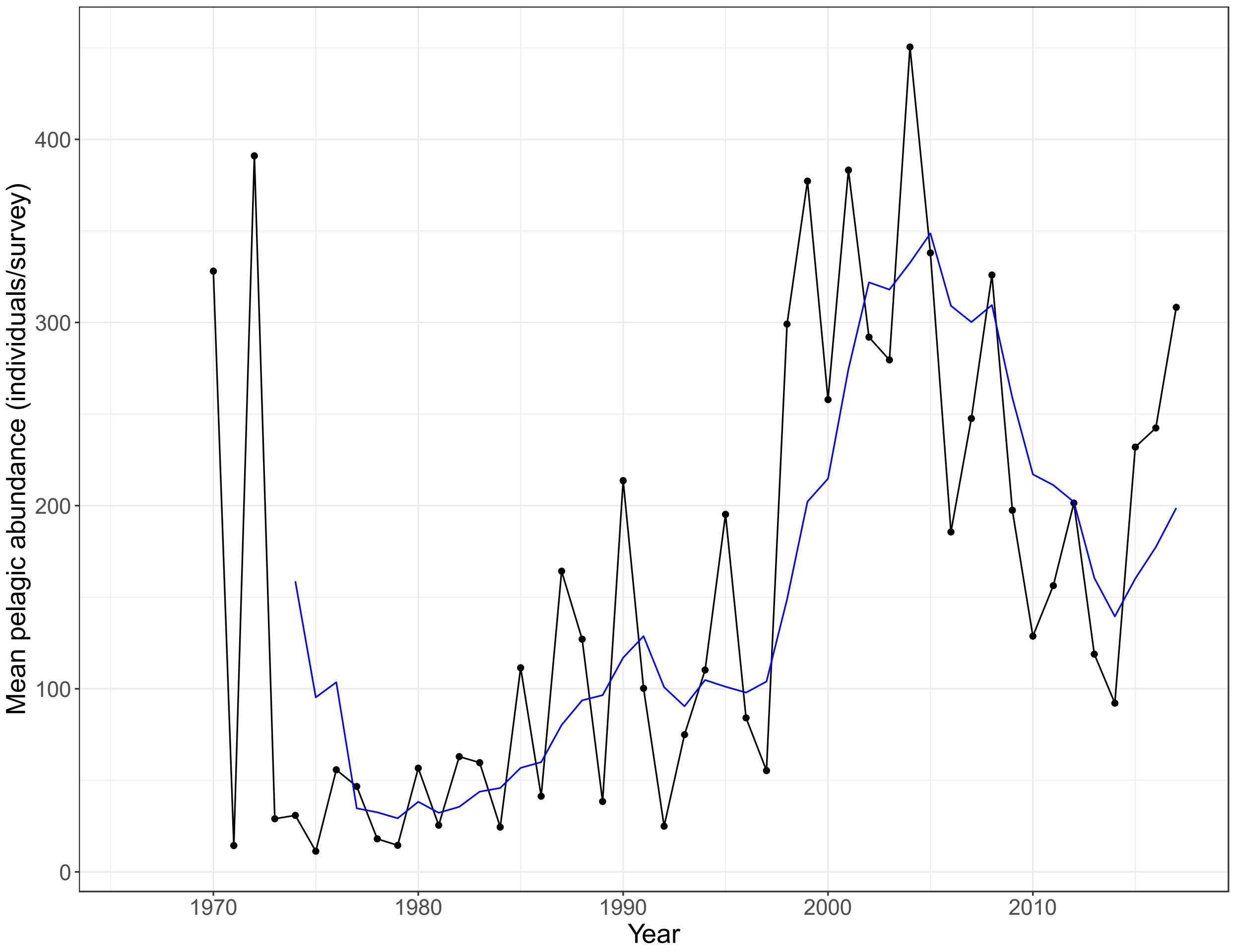

Figure A.3: Temporal trend of mean pelagic abundance (individuals/survey) on the Eastern Scotian Shelf from 1970 to 2017. Blue line represents the five-year moving average.

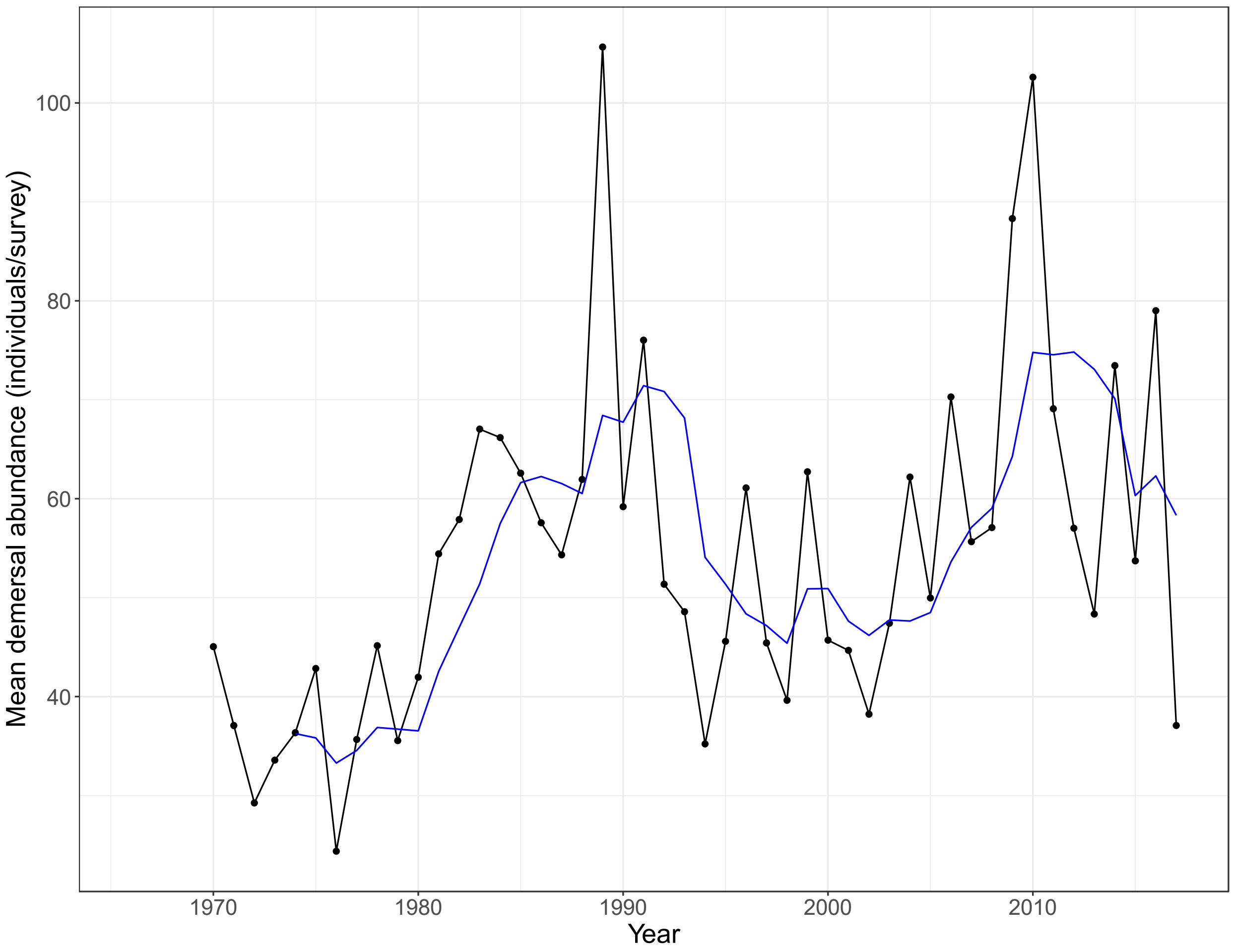

Figure A.4: Temporal trend of mean demersal abundance (individuals/survey) on the Eastern Scotian Shelf from 1970 to 2017. Blue line represents the five-year moving average.

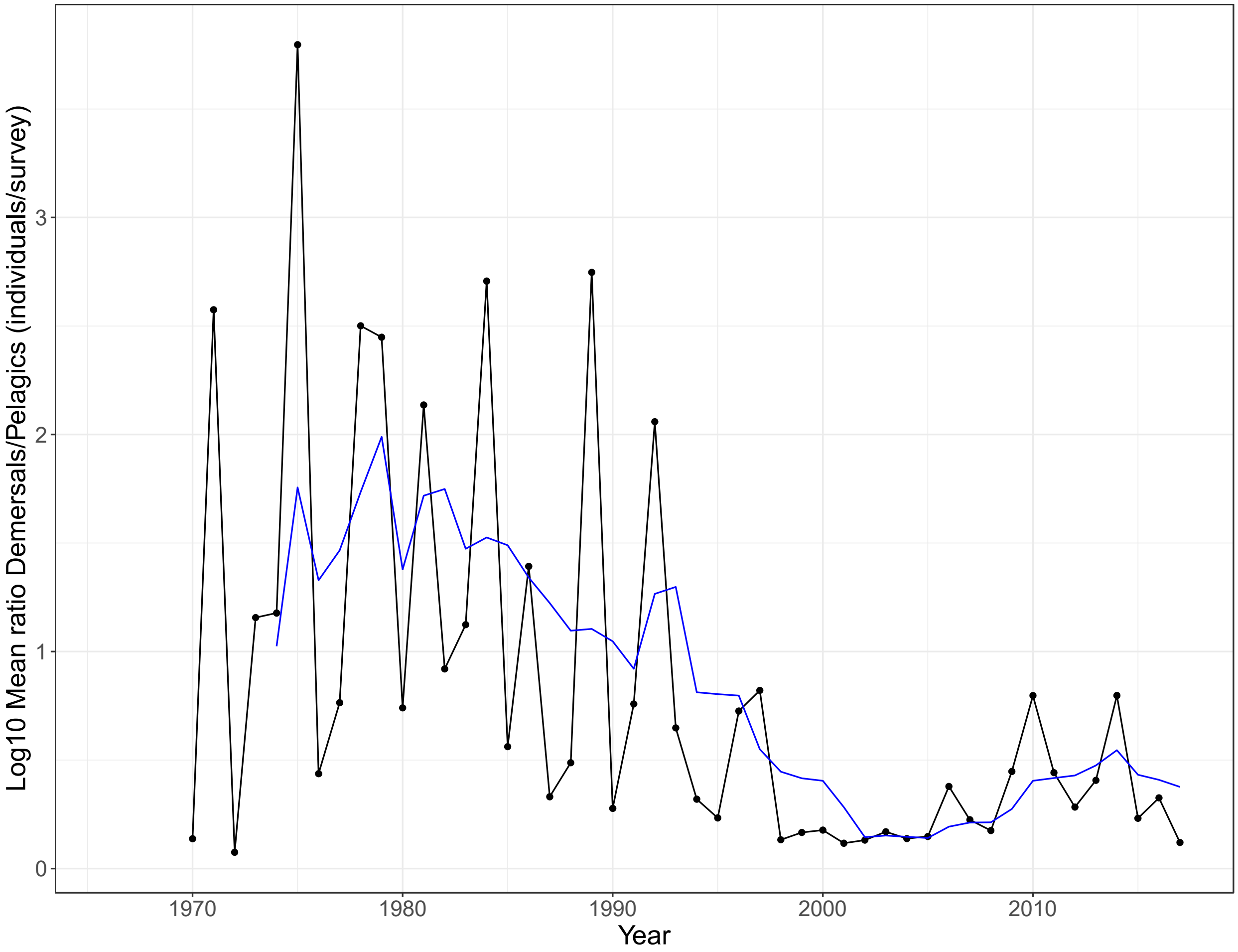

Figure A.5: Temporal trend of the ratio of mean demersal to pelagic abundance (individuals/survey) on the Eastern Scotian Shelf from 1970 to 2017. Blue line represents the five-year moving average.

Table A.1: Reproducible framework for performing Red List of Ecosystems assessments of offshore marine ecosystems.

Criteria	Subcriteria	Suggested approach	Data requirements
Criterion A	A1	Apply fuzzy clustering to determine boundaries and spatial extent of the ecosystem. Classify focal ecosystem and surrounding ecosystems into separate clusters at the beginning of the time period, based on their characteristic biological community. Then identify clusters at the end of the time period to assess whether there has been any spatial decline or shift in the ecosystem.	Timeseries of ~50 years of species abundance in the ecosystem of interest and at least one neighbouring ecosystem.
	A2a	DD	DD
	A2b	DD	DD
	A3	DD	DD
Criterion B	B1	Determine the total area of the ecosystem (km2)	Description of ecosystem in the literature.
	B2	Calculate the number of 10 x 10 km grid cells, by overlaying a shapefile of the ecosystem with a 10 x10 km grid.	Shapefile of ecosystem
	B3	Estimate number of separate patches/areas within ecosystem and risk of threats in near future (~20 years)	Description of ecosystem in the literature.
Criterion C	C1	Using information in the literature, set collapse threshold values for chosen abiotic indicators. Examine the timeseries of chosen abiotic indicators to determine whether the collapse threshold has been crossed.	Collapse thresholds for abiotic indicators. Timeseries of ~50 years of chosen abiotic indicators in the ecosystem of interest. Potential indicators include sea surface temperature, bottom temperature and nutrient concentrations.

	C2a	Assess the projected abiotic indicators from the ecosystem model (see Criterion E) to determine whether the collapse threshold identified under subcriterion C1 has been crossed.	Timeseries of ~50 years of abiotic indicators from projected ecosystem run (see Criterion E).
	C2b	DD	DD
	C3	DD	DD
Criterion D	D1	Calculate the percentage change in the chosen biotic indicators between the beginning and end of the timeseries.	Timeseries of ~50 years of chosen biotic indicators in the ecosystem of interest. Potential indicators include trends in key functional groups (e.g. demersals), and ratios of key functional groups (e.g. demersals:pelagics),
	D2a	DD	DD
	D2b	DD	DD
	D3	DD	DD
Criterion E		Project the ecosystem under climate change and fishing scenarios for ~100 years using an ecosystem model capable of examining the physiological impacts of temperature and fishing, such as "*therMizer*".	Timeseries of ~50 years of spawning stock biomass estimates, fishing mortality, and catch or landings data for each species. Species-specific life history parameters from the literature. Projected sea surface temperature, bottom temperature and plankton concentration from an Earth System Model, under a climate change scenario (e.g. RCP 8.5).

APPENDIX B: SUPPLEMENTARY MATERIALS FOR CHAPTER 4

Table B.1: Length-weight conversions for species included in the ecosystem model. Maximum lengths (*Lmax*) are converted to weights following the standard exponential equation $W = a.Lmax^{b}$. References for *a*, *b* and *Lmax* are found in the species column.

Species	Lmax	Length unit	a	b
American Plaice (*Hippoglossoides platessoides*)[a]	59	cm	0.004	3.176
Atlantic cod (*Gadus morhua*) [a]	121	cm	0.00689	3.076
Haddock (*Melanogrammus aeglefinus*) [a]	69	cm	0.00849	3.034
Atlantic herring (*Clupea harengus*) [b]	346	mm	0.000002	3.21
Redfish (*Sebastes sp.*) [a]	61	cm	0.01057	3.114
Spiny dogfish (*Squalus acanthias*) [a]	99	cm	0.00147	3.25
Turbot, Greenland halibut (*Reinhardtius hippoglossoides*) [a]	74	cm	0.0035	3.245
Witch flounder (*Glyptocephalus cynoglossus*) [a]	57	cm	0.00249	3.249
Yellowtail flounder (*Pleuronectes ferruginea*) [a]	44	cm	0.00749	3.018

[a] Noble & Clark, 2019a
[b] Noble & Clark, 2019b

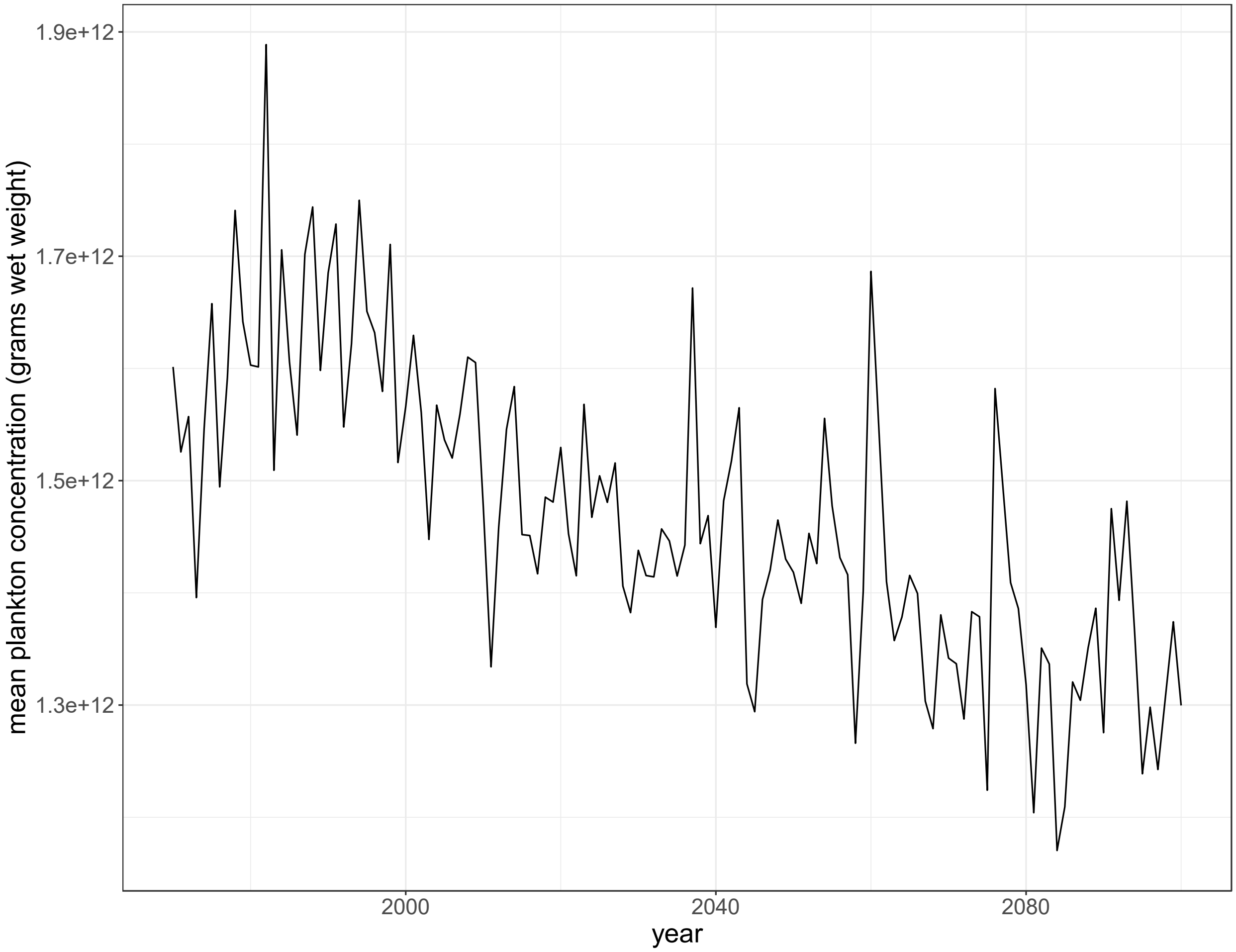

Figure B.1: IPSL projected plankton concentration on the Eastern Scotian Shelf under RCP 8.5. from 1970-2100.

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