AMAP Assessment 2018: Arctic Ocean Acidification

AMAP

Arctic Monitoring and Assessment Programme (AMAP)

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Preface

This assessment report presents the results of the 2018 Arctic Monitoring and Assessment Programme (AMAP) assessment of Arctic Ocean Acidification. This is the second such assessment dealing with ocean acidification from an Arctic-wide perspective, and complements several assessments that AMAP has delivered over past years concerning the effects of climate change on Arctic ecosystems and people.

The main objectives of this assessment were to:

- Update the understanding of chemical and biological responses to ocean acidification since the first Arctic Ocean acidification assessment in 2013.
- Evaluate how ecological frameworks may respond to ocean acidification.
- Research socioeconomic and cultural consequences of ocean acidification at selected case study regions.
- Report on how Arctic Ocean acidification may impact on downstream global systems.
- Deliver guidance for management of change.

AMAP is a group working under the Arctic Council. The Arctic Council Ministers have requested that AMAP:

- Produce integrated assessment reports on the status and trends of the conditions of the Arctic ecosystems.
- Identify possible causes for the changing conditions.
- Detect emerging problems, their possible causes, and the potential risk to Arctic ecosystems including indigenous peoples and other Arctic residents.
- Recommend actions required to reduce risks to Arctic ecosystems.

This assessment report provides the scientific basis and validation for the statements and recommendations made in the *AMAP Arctic Ocean Acidification: Assessment Summary for Policy-makers* that will be delivered to Arctic Council Ministers at their meeting in Rovaniemi, Finland in May 2019. It includes extensive background data and references to the scientific literature. The summary for policy-makers report contains recommendations that focus mainly on policy-relevant actions concerned with addressing the consequences of ocean acidification in the Arctic, while the conclusions and recommendations presented in this report cover issues of a more scientific nature, such as proposals for filling gaps in knowledge, and recommendations relevant to future monitoring and research work.

This assessment was conducted between 2014 and 2018 by an international group of over 40 experts. Lead authors were selected based on an open nomination process coordinated by AMAP. A similar process was used to select international experts who independently reviewed this report.

Information contained in this report is fully referenced and based, first and foremost, on peer-reviewed and published results of research and monitoring. It also incorporates some new (unpublished) information from monitoring and research conducted according to well established and documented national and international standards and quality assurance / quality control protocols. Care has been taken to ensure that no critical statements are based on non-peer-reviewed materials.

Access to reliable and up-to-date information is essential for the development of science-based decision-making regarding ongoing changes in the Arctic and their global implications. The *AMAP Arctic Ocean Acidification: Assessment Summary for Policy-makers* has therefore been developed specifically for policy-makers, summarizing the main findings of this science-based report. The assessment lead authors have confirmed that both this report and its derivative products accurately and fully reflect their scientific assessment. The reports are freely available from the AMAP Secretariat and on the AMAP website: www.amap.no, and their use for educational purposes is encouraged.

AMAP would like to express its appreciation to all experts who have contributed their time, efforts and data, in particular the lead authors who coordinated the production of this report. Thanks are also due to the reviewers who contributed to the peer-review process and provided valuable comments that helped to ensure the quality of the report. A list of the main contributors is included at the start of each chapter. The list is not comprehensive. Specifically, it does not include the many national institutes, laboratories and organizations, and their staff, which have been involved in various countries in Arctic ocean acidification-related monitoring and research. Apologies, and no lesser thanks are given to any individuals unintentionally omitted from the list.

The support from the Arctic countries and non-Arctic countries implementing research and monitoring in the Arctic is vital to the success of AMAP. The AMAP work is essentially based on ongoing activities within these countries, and the countries that provide the necessary support for most of the experts involved in the preparation of the AMAP assessments. In particular, AMAP would like to acknowledge Norway for taking the lead country role in this assessment and to thank Canada, Denmark, Germany, Norway, Sweden, USA and the Nordic Council of Ministers for their financial support to the assessment work.

The AMAP Working Group is pleased to present its assessment to the Arctic Council and the international science community.

Richard Bellerby (Assessment Chair)

Marianne Kroglund (AMAP Chair)

Rolf Rødven (AMAP Executive Secretary)

Tromsø, October 2018

Disclaimer: The views expressed in this peer-reviewed report are the responsibility of the authors of the report and do not necessarily reflect the views of the Arctic Council, its members or its observers.

1. Introduction

Author: Richard Bellerby

Ocean acidification, resulting from changes in ocean chemistry induced by increasing seawater carbon dioxide concentrations, is one of the growing challenges to marine organisms, ecosystems and biogeochemical cycling. Some of the fastest rates of ocean acidification currently observed are in the Arctic Ocean, with important physiological and geochemical thresholds already surpassed. Projections indicate that large parts of the Arctic Ocean are undergoing marine carbonate system changes that will incur significant shifts in ecological status over the coming decades unless global carbon emissions are drastically curtailed. These changes in water chemistry and biology will have significant socio-ecological and economic consequences at the local to global level.

The first AMAP Arctic Ocean acidification report (AMAP, 2013) presented a scientific assessment on the changing state of ocean acidification in the Arctic and provided an Arctic-wide perspective on the rapid increase in seawater acidity. The report concluded that ocean acidification was affecting the Arctic marine environment and ecosystems. Key findings were as follows:

- Arctic marine waters are experiencing widespread and rapid ocean acidification with the primary driver being uptake of carbon dioxide emitted to the atmosphere by human activities.
- The Arctic Ocean is especially vulnerable to ocean acidification, which is not uniform across the Arctic Ocean.
- Arctic marine ecosystems are highly likely to undergo significant change due to ocean acidification, with direct and indirect effects on Arctic marine life.
- It is likely that some marine organisms will respond positively to new conditions associated with ocean acidification, while others will be disadvantaged, possibly to the point of local extinction.

The 2013 AMAP report further emphasized that ocean acidification impacts must be assessed in the context of other changes happening in Arctic waters since ocean acidification is one of several factors (stressors) that may contribute to alteration of fish species composition and abundance in the Arctic Ocean. Ecosystem changes associated with ocean acidification and other stressors may affect the livelihoods of Arctic peoples. AMAP (2013) recommended that the Arctic Council adopt the following actions:

- Urge Member States, observer countries, and global society to reduce carbon dioxide emissions as a matter of urgency.
- Enhance research and monitoring efforts that expand our understanding of acidification processes and the effects on Arctic marine ecosystems and northern societies that depend on them.
- Advise Member States to implement adaptation strategies that address all aspects of Arctic change, including ocean acidification, tailored to local and societal needs.

Based on the key findings and ensuing Arctic Council recommendations, it was decided to build on the AMAP 2013 assessment by developing regional case studies. These case studies allow for the evaluation of selected impacts of the progression of Arctic Ocean acidification within the context of other environmental changes that affect local communities and regional to global economies. The Arctic ministers requested, in particular, consideration of potential ecological, socio-economic and global implications of ocean acidification in this new report, including global teleconnections.

General chemistry and physiological background of ocean acidification were described in the previous report (AMAP, 2013). This report reviews recent new understanding on regional Arctic Ocean acidification (Chapter 2), biological responses to ocean acidification (Chapter 3) and provides a general overview of some socio-economic impacts (Chapter 4). The legal framework of Arctic Ocean acidification is examined in Box 1.1 by use of examples of global and regional laws and policies relevant to addressing the potential impacts of climate change and ocean acidification. The report concludes with six annexes, the first covering ocean acidification conditions in the waters entering the Arctic Ocean and how these are affected by processes taking place within the Arctic Ocean itself (Annex 1). The remaining five annexes provide detailed information on regional ocean acidification specific to particular locations/species, and deliver end-to-end representation of systems that integrate key biogeophysical and socio-economic processes. These cover urchins and kelp in Norway (Annex 2), ocean services for the Barents Sea (Annex 3), shrimp in Greenland (Annex 4), risk assessment of Alaska's fishery sector (Annex 5), and changing ocean impacts on the key forage fish species Arctic cod in the Western Canadian Arctic (Annex 6).


Knud Falk

Box 1.1 Examples of global and regional laws and policies relevant to addressing the potential impacts of climate change and ocean acidification

Authors: David L. VanderZwaag, Nadja Steiner, Jan René Larsen

Laws and policies relevant to the potential impacts of climate change and ocean acidification on marine species and coastal communities appear at the global and regional level, as well as the national level (see Annex 6).

Global

While a broad range of international agreements and texts are relevant to ocean acidification (Stephens, 2015; Fennel and VanderZwaag, 2016), the Paris Agreement continues to set the overall agenda for advancing mitigation and adaptation responses to climate change and ocean acidification (Klein et al., 2017). This Agreement establishes an overall objective of keeping the global average temperature rise this century to well below 2°C above pre-industrial levels and to pursue efforts to limit the temperature increase even further to 1.5°C above pre-industrial levels. A key to mitigation is the requirement for Parties to submit nationally determined contributions (NDCs) which set out planned domestic mitigation measures; successive NDCs are expected to become increasingly progressive and ambitious (Winkler, 2017). Article 7 specifies the Parties' adaptation responsibilities (Pérez and Kallhauge, 2017).

How good the Paris commitments will be for addressing ocean acidification is unclear. The Paris Agreement focuses on constraining the global average rise in air temperature without any specific consideration to controlling ocean pH levels, and substantial scientific uncertainty continues to exist concerning what the temperature target means for the oceans, and ocean acidification in particular (Harrould-Kolieb, 2016). The Intergovernmental Panel on Climate Change is due to publish two special reports which may help to increase understanding of ocean acidification: one in October 2018 on the potential impacts of the 1.5°C target for human populations and natural ecosystems (IPCC, 2017a) and one in 2019 on climate change impacts on the oceans and cryosphere (IPCC, 2017b).

Two other global initiatives stand out. First, the United Nations 2030 Agenda for Sustainable Development, specifically Goal 14 on oceans, sets a target of minimizing and addressing the impacts of ocean acidification, including through enhanced scientific cooperation at all levels (UN General Assembly, 2015). Second, in April 2018, the International Maritime Organization adopted an initial strategy on the reduction of greenhouse gas (GHG) emissions from ships. The strategy calls for a peak in GHG emissions from international shipping as soon as possible and a reduction in the total annual GHG emissions of at least 50% by 2050 compared to 2008, while pursuing efforts towards phasing them out (IMO, 2018).

Climate change and ocean acidification impacts in the Arctic threaten various internationally recognized human and Indigenous rights (Duyck, 2015; de Windt et al., 2016). These rights, some set out in the UN Declaration on the Rights of Indigenous Peoples (2007) include, among others, the rights to food, subsistence, culture and human health (McCrimmon, 2016; Special Rapporteur, 2017).

Regional

In November 2017, the five Arctic Ocean coastal States (USA, Canada, Russia, Norway, and Denmark in respect of the Faroe Islands and Greenland), as well as the EU and four other States (China, Iceland, Japan, and the Republic of Korea) successfully concluded negotiations on a draft Agreement to Prevent Unregulated High Seas Fisheries in the Central Arctic Ocean (U.S. Department of State, 2017). The Agreement, adopting a precautionary approach to potential future commercial fisheries, commits Parties to establish a Joint Program of Scientific Research and Monitoring. Parties agree not to authorize their flagged vessels to conduct commercial fishing in the Agreement Area until conservation and management measures are adopted by one or more regional or subregional management organizations or arrangements or interim measures are adopted by the Parties pursuant to the Agreement.

Actions of the Arctic Council's Arctic Marine Strategic Plan 2015-2025 (PAME, 2015a) include strengthening the collection and monitoring of data, as well as more communicative measures to enhance understanding of the consequences of ocean acidification. The Arctic Council has been active in encouraging adaptation efforts in the Arctic. It has adopted a Framework for a Pan-Arctic Network of Marine Protected Areas (MPAs) aiming to develop networks of MPAs within national jurisdictions of Arctic States and seeking to improve resilience to climate change and other hazards (PAME, 2015b). Pursuant to the Framework, a further guidance document on other area-based conservation measures, such as seasonal closures to protect a particular species, has also been published (PAME, 2017).

The Baltic Marine Environment Protection Commission (HELCOM) is an intergovernmental organization governing the Convention on the Protection of the Marine Environment of the Baltic Sea Area (Helsinki Convention). In the 2018 Ministerial Declaration, Parties commit to increase the protection and restoration of biodiversity and to preserve and promote the ecological balance of the Baltic Sea, also as a response to adaptation needs stemming from human-induced climate change (HELCOM, 2018).

Through the Convention for the Protection of the Marine Environment of the North-East Atlantic (OSPAR Convention), the OSPAR Commission has worked to identify threats to the North-East Atlantic and has organized programs and measures to ensure effective national action to combat them. OSPAR focuses on the challenges that human activities pose for marine ecosystems, such as pollution, eutrophication, and climate change. OSPAR seeks to monitor and assess the rate and extent of the effects of ocean acidification, and to consider appropriate ways of reducing these impacts (OSPAR, 2010; ICES, 2014).

2. Arctic Ocean acidification: an update

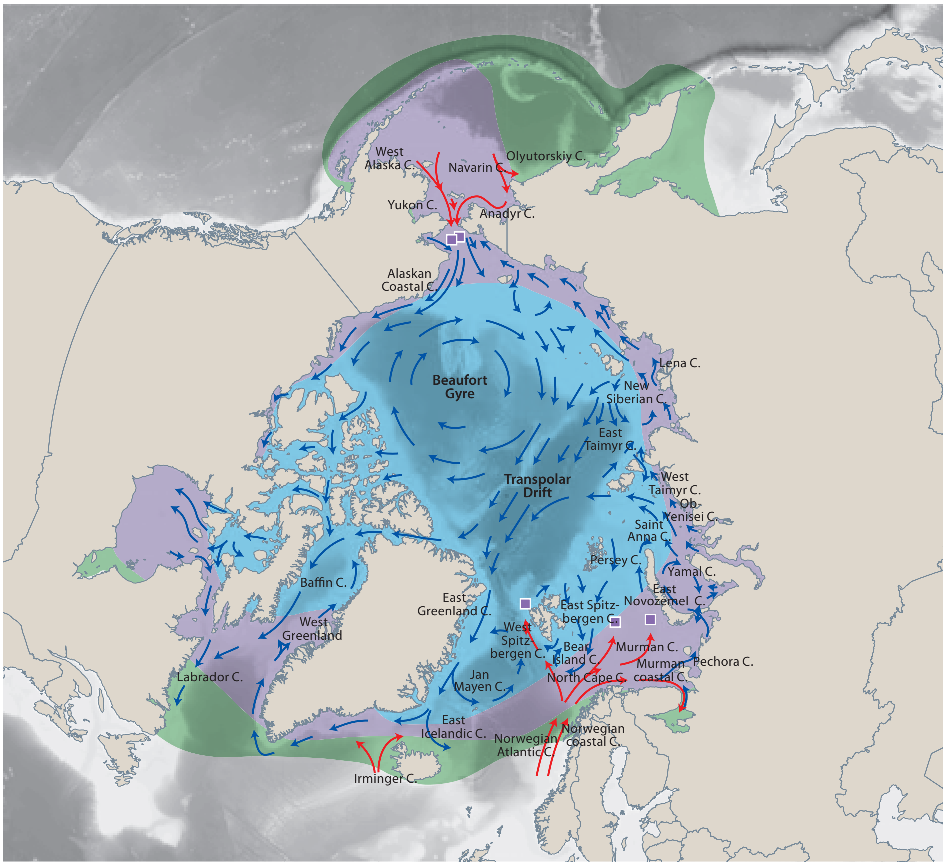
Authors: Richard Bellerby, Leif G. Anderson, Emily Osborne, Nadja Steiner, Irina Pipko, Jessica Cross, Melissa Chierici, Agneta Fransson, Kumiko Azetsu-Scott, Jón Ólafsson, Lisa Miller

2.1 Introduction

This chapter reports on recent advances in our understanding of ocean acidification from a chemical or marine carbonate system perspective. It builds on the last AMAP report on ocean acidification (AMAP, 2013) and follows the prior example of discussing the specific Arctic regions individually, where sufficient information allows. Ocean acidification refers to the reduction in the pH of the ocean over an extended period, typically decades or longer, which is caused primarily by uptake of carbon dioxide (CO_2) from the atmosphere, but can also be caused by other chemical additions or subtractions from the ocean. Anthropogenic ocean acidification refers to the component of pH reduction that is caused by human activity (IPCC, 2011; AMAP, 2013). In the Arctic, ocean acidification is intensified due to low temperatures, increased freshwater supply (river runoff and ice melt) and low pH Pacific water inflow. Seawater pH, the partial pressure of CO_2 (pCO_2) and the saturation states of aragonite (Ω_{arg}) and calcite (Ω_{cal}) are commonly used as measures for the status of acidification. They are major drivers of key marine physiological processes, and are used to indicate potential challenges to some marine species (see Chapter 3). A more detailed review of marine carbonate chemistry, ocean acidification and the key processes controlling carbon cycling in the Arctic can be found in the first AMAP Ocean Acidification assessment report (AMAP, 2013). Additionally, a study of the export of ocean acidification from the Arctic ocean to the North Atlantic can be found in Annex 1.

To follow the sources and fate of carbon into and around the Arctic Ocean, this chapter tracks the general circulation pattern in Figure 2.1 discussing the changing marine carbonate system and identifying causes of ocean acidification, where possible, in the different regional seas and basins. There are two inflows to the surface Arctic Ocean via the Atlantic and Pacific gateways (Figure 2.2). The former has the greatest influence on water mass structure and currents, thus the Atlantic inflow is addressed first, while contributions from the Pacific and meteoric sources have strong local to regional influences.

From a starting point in the northern section of the North Atlantic Current, commonly termed the Norwegian Atlantic Current, the uptake of anthropogenic CO_2 from the atmosphere dominates in the water flowing north from the Atlantic Ocean into the Arctic Ocean. Some of this water penetrates to intermediate depths (a few kilometers) when the warm salty water cools, mainly in the Barents Sea, before entering through the St. Anna Trough into the northern Kara



🟦 High Arctic
🟪 Low Arctic
🟩 Subarctic
→ Warm current
→ Cold current

Figure 2.1 Major surface ocean currents within the High Arctic, Low Arctic and subarctic areas of the Arctic Ocean and its marginal seas.

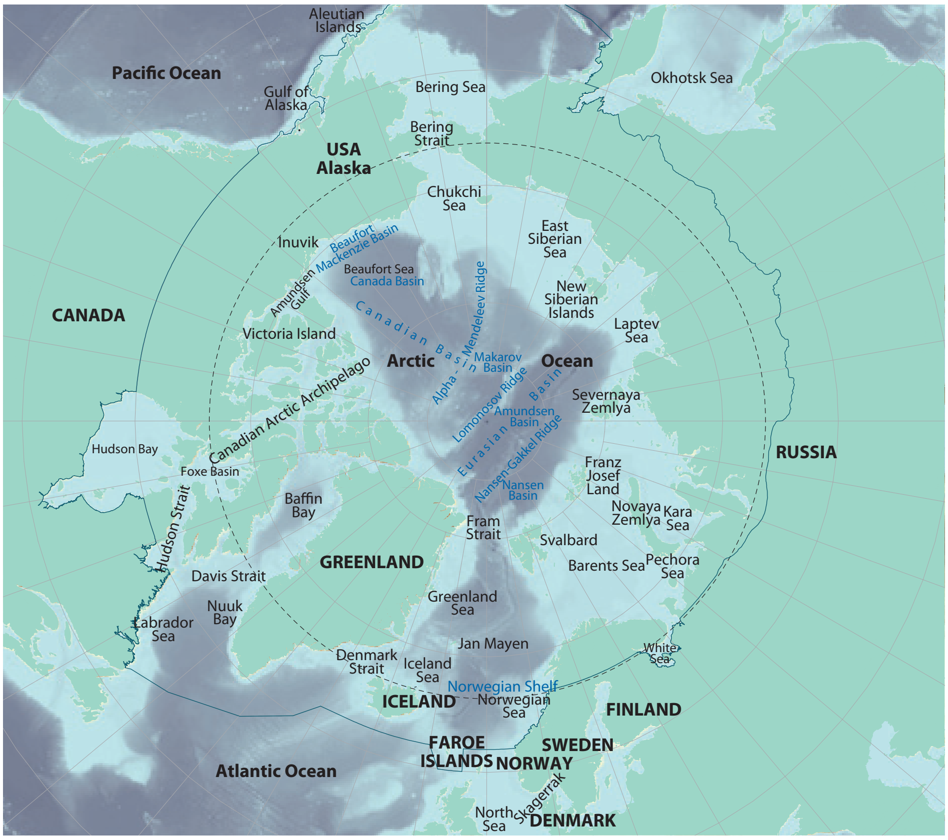


Figure 2.2 The Arctic Ocean and its marginal seas.

Sea. This inflowing water follows the continental margin towards the east and, influenced by ocean ridges, forms the large-scale cyclonic (anti-clockwise) gyres around the deep central basins.

In some coastal and shelf regions, high concentrations of CO_2 result from degradation of organic matter; a feature that is especially relevant in the Laptev Sea, East Siberian Sea and Chukchi Sea. Here, large amounts of organic matter, both dissolved and particulate, is supplied from land through river runoff and coastal erosion. Further to the east is found high marine primary production, stimulated by the inflow of high nutrient water from the Pacific Ocean. The majority of the organic carbon is introduced in particulate form. Dissolved organic carbon (DOC) fluxes from coastal erosion are low compared to particulate (and total) organic carbon fluxes, and play a secondary role in the Arctic carbon budget. However, this DOC is assumed to be highly bioavailable, and it could be important for ecosystems in the Arctic nearshore zones, particularly in summer when river discharge is low, and in areas where rivers are absent (Fritz et al., 2015; Tanski et al., 2016). In the East Siberian Sea and Chukchi Sea, high CO_2 concentrations are most obvious in the bottom waters to where the majority of organic matter is exported – near the ocean-sediment interface prior to degradation. Strong pelagic-benthic coupling, with export percentages of over 50% of fixed organic carbon, is for example common across the eastern Chukchi Sea shelf (Strong et al., 2016). This water is high in CO_2, as well as in nutrients and flows off the shelf into Canada Basin where it is 'trapped' for some time in the Beaufort Gyre centered around a depth of about 125 m. Water leaves the Beaufort Gyre along the continental slope north of the Canadian Arctic Archipelago and Greenland and flows into the north Atlantic following currents that straddle Greenland. Here water high in CO_2 is promoted closer to the surface when the overlying low salinity surface water is mixed with surrounding higher salinity water. The horizontal extent of the Beaufort Gyre varies with time depending on the dominating atmospheric pressure field, leading to large fluctuations in the magnitude of the outflow of the high CO_2 water. Surface water of the Beaufort Gyre also contains low salinity water, originating from both sea ice melt and river runoff, both of which have increased with climate warming. Freshening of surface seawater that is in equilibrium with atmospheric CO_2 results in a lower pH, if the total alkalinity decreases with salinity, thus also contributing to ocean acidification. Furthermore, freshening also lowers the concentration of calcium ions and therefore further lowers the saturation state of calcium carbonate minerals. Due to ice formation and melt in the Arctic Ocean at certain times of the year, sea-ice processes such as CO_2-rich brine rejection and calcium carbonate ($CaCO_3$; ikaite) precipitation/dissolution affect the carbonate chemistry in underlying water (e.g., Rysgaard et al., 2007, 2012, 2013; Fransson et al., 2013, 2017; Miller et al., 2014; Geilfus et al., 2016). Results indicate that ikaite export from sea ice and its dissolution in the underlying seawater can potentially reduce the degree of oceanic acidification in autumn and in winter in ice-covered areas, at the time when Ω_{arg} is lowest. This process has been suggested in several studies such as in Young Sound Greenland (Rysgaard et al., 2012), in the Canadian Arctic Archipelago (Fransson et al., 2013), and in the Nansen Basin (Fransson et al., 2017), and further confirmed through an experiment in Young Sound (Rysgaard et al., 2014; Geilfus et al., 2016).

2.2 Marine carbonate system and ocean acidification in the regional seas and basins

2.2.1 The Western Eurasian Shelves

2.2.1.1 Barents Sea

Ocean acidification in the Barents Sea is influenced by the variability in the inflowing waters such as the fresh coastal waters, the warm waters transported with the North Atlantic Current and the inflow of cold and fresh waters from the Arctic. Moreover, carbon cycling, and thus ocean acidification, in the northern parts are also influenced by sea-ice cover, moderating air-sea CO_2 exchange, and internal ice processes during formation and melting (Nedashkovsky and Shvetsova, 2010; Rysgaard et al., 2012; Fransson et al., 2017). Other important mechanisms for atmospheric CO_2 uptake are cooling of the warm water transported by the North Atlantic Current and deep-water formation such as in Storfjorden (Anderson et al., 2004). In the Barents Sea Opening, organic matter production and decay play leading roles in the carbonate system seasonal dynamics (Yakushev and Sørensen, 2013). As observed during 2010, seasonal variations in surface pH were about 0.2, and the aragonite saturation values were in the range 1.4–2.6, with a minimum in February near the Spitsbergen coast and a maximum in the same region in May. Observations over an annual cycle of the carbonate system of surface waters in the Atlantic gateway to the Arctic Ocean, covering the region between Svalbard and mainland Norway, also revealed significant seasonal variability in Ω_{arg} (0.4–0.9), with the lowest saturation states recorded in winter ($\Omega_{arg} \sim 1.8–2.1$) and the highest recorded in spring and summer ($\Omega_{arg} \approx 2.4$) (Tynan et al., 2014). The surface water pCO_2 in winter is generally undersaturated (mean of 370 µatm) relative to the atmospheric level of 400 µatm, except for occasional oversaturation in West Spitsbergen fjords. Biological CO_2 uptake further decreases the surface water pCO_2 by 100 µatm from winter to summer, with the greatest undersaturation occurring in the north (Chierici et al., 2017; Pipko et al., 2017). Fransson et al. (2001) estimated that about 70% of the oceanic CO_2 uptake was caused by biological CO_2 uptake. In the Barents Sea Opening, pH values are lowest in the bottom waters (8.04–8.05) at 71.5°N. Aragonite saturation states decrease northwards, from 1.8–2.0 in the upper 200 m until the polar front at 73.5°N, where Ω_{arg} reaches lowest values of 1.5 near Bjørnøya, caused by the influence of fresher polar water from the North. Surface water pH and Ω_{arg} were 8.06–8.07 and 1.5–1.8, respectively, with the highest values found in summer (8.11–8.15 and 1.9–2.1). In contrast, the Barents Sea Opening was supersaturated with respect to aragonite throughout the year.

There are very few estimates of the anthropogenic CO_2 content in the Barents Sea and even fewer data to assess ocean acidification trends in the Barents Sea (AMAP, 2013), which is probably due to large variability in the water column and large data gaps. From 1967 to 2001, surface water pCO_2 increased by 42±31 µatm (Omar et al., 2003), tracking the atmospheric CO_2 increase. Analyzing a longer period from 1981 to 2013, Skjelvan et al. (2014) were unable to detect a significant trend in ocean acidification and they attributed this to the high seasonal variability of the coupled climate-marine carbonate system. This was also the case for the period 2011 to 2016, where no clear trend in Ω_{arg} was found in the Barents Sea Opening (Chierici et al., 2017). The northern and eastern part of the Barents Sea has recent data coverage but too few data to assess a trend (Chierici et al., 2017).

2.2.1.2 Svalbard fjords

The Svalbard fjords are strongly influenced by seasonal and interannual variability in glacial water runoff and its effect on the carbonate chemistry, where increased freshwater decreases pH and Ω (Fransson et al., 2015, 2016). In fjords such as Kongsfjorden and Tempelfjorden, Ω_{arg} is close to critical limits <1.4 (Fransson et al., 2015, 2016) for the pteropod *Limacina helicina* to form aragonite shells (Bednaršek et al., 2012, 2014). The decrease in Ω due to glacial meltwater is mostly counteracted by the increase due to biological CO_2 consumption. In addition, in some Svalbard fjords, such as Tempelfjorden and Kongsfjorden which are affected by tide-water glaciers, the glacial drainage water contributes to alkalinity due to additional carbonate ions from minerals in the bedrock (e.g., limestone, calcite and dolomite), which may partly mitigate ocean acidification (Fransson et al., 2015, 2016). During years with larger freshwater inputs, there was a larger effect of bedrock-originated calcium carbonate ions, which increased total alkalinity, particularly in the inner parts of the fjords (Fransson et al., 2015, 2016). In Storfjorden, there is a polynya where ice formation and deep-water formation are taking place. When sea ice is formed, dense, high-salinity CO_2-rich brine is produced and rejected from the ice to underlying water (e.g., Fransson et al., 2013). This sinking of brine with CO_2 has been shown to increase CO_2 and dissolved inorganic carbon (C_T) hence decreasing Ω and pH in the bottom waters of the fjord (Anderson et al., 2004).

2.2.1.3 North of Svalbard: the Nansen Basin and shelf slope

The area north of Svalbard and the Nansen Basin shows strong seasonal variability in carbonate chemistry of the surface water under the sea ice from January to June (Fransson et al., 2017). The fugacity of CO_2 (fCO_2) in the surface water was shown to be undersaturated in relation to atmospheric fCO_2 during all seasons. This was mainly due to ikaite dissolution from sea-ice processes from winter to spring and biological CO_2 consumption in spring. Although sea-ice cover is dominant in winter, openings in the ice cover due to large storms can facilitate ocean CO_2 uptake (Fransson et al., 2017). The net CO_2 sink varied from 0.3 to 86 mmol C/m^2/d, depending strongly on the open-water fractions and storm events. The relative effects of the total carbonate chemistry changes were estimated to be from $CaCO_3$ dissolution (38%), biological CO_2 consumption (26%), vertical mixing (16%), and air-sea CO_2 fluxes (16%); temperature and salinity contributions were judged to be insignificant (Fransson et al., 2017).

2.2.1.4 Kara, Laptev and East Siberian Seas

The water entering the Kara Sea and Laptev Sea is transported by currents that have transgressed the Barents Sea and an easterly flow of the West Spitzbergen Current extension.

Surface currents follow the continental shelf and slope and, while some intermediate water formation transfers CO_2 to depth, the remainder continues along the shelf and shelf slope to the east. In summer, a gradual increase in CO_2 concentration from west to east from the Barents Sea to Laptev Sea results from the varying influences of riverine inputs of freshwater and organic carbon (Pipko et al., 2017). Generally, the influence of riverine inputs on pCO_2 increases to the south in both the Kara Sea and Laptev Sea (Pipko et al., 2017). The high pCO_2 values found in the summertime Laptev Sea, outside the Lena river delta result from high terrestrial organic carbon decay (Semiletov et al., 2011, 2013).

The extensive East Siberian Arctic Shelf, made up of the Laptev Sea, the East Siberian Sea, and the Russian part of the Chukchi Sea, which comprises ~25% of the Arctic continental shelf, is a particularly vulnerable area to ocean acidification. It receives riverine discharge from four large Arctic rivers. Moreover, the near-shore system, including riverbanks, deltas and coastlines, includes thousands of kilometers of Pleistocene ice complexes with massive ice wedges and large amounts of organic carbon. This system is strongly affected by warming and the East Siberian Arctic Shelf can be considered an integrator of ongoing changes in the surrounding environment (Semiletov et al., 2013, 2016). Two biogeochemical provinces were identified on the East Siberian Arctic Shelf (Semiletov et al., 2005). The warmer and fresher Western Biogeochemical Province water is characterized by strong river and coastal erosion impacts. Moreover, high concentrations of riverine chromophoric dissolved organic matter affect both primary production and ultraviolet exposure in this area (Pugach et al., 2018). The saltier and colder Eastern Biogeochemical Province water is mainly affected by nutrient-rich Pacific water that creates favorable conditions for high summer primary production.

The lowest Ω_{arg} levels were observed in the Western Biogeochemical Province, where the influence of both river runoff and terrestrial organic carbon input is much stronger than in the Eastern Biogeochemical Province (Figure 2.3). In the Western Biogeochemical Province in summer, Ω_{arg} varied from 0.01 to 1.42 (mean 0.45) in surface water (above the pycnocline) and from 0.01 to 1.27 (mean 0.44) in bottom water (below the pycnocline). It has been found that waters of the Laptev Sea inner shelf throughout the water column and surface and bottom waters of the middle shelf are strongly undersaturated with both aragonite and calcite (Pipko et al., 2015). The calcium carbonate saturation state of the shallow Laptev Sea shelf water in the summer-autumn seasons has significant interannual variability, driven mainly by wind strength and direction and by the volume of the terrestrial (river and erosional) runoff. Surface waters of the Laptev Sea inner and middle shelves are a strong seasonal source of CO_2 to the atmosphere; whereas surface waters of the outer shelf and continental slope in general are a sink for atmospheric CO_2 in the summer-autumn seasons (Pipko et al., 2016). However, recently the heterotrophic area has expanded to the outer shelf and continental slope, and their surface waters can also be a source of CO_2 to the atmosphere (Pipko et al., 2017).

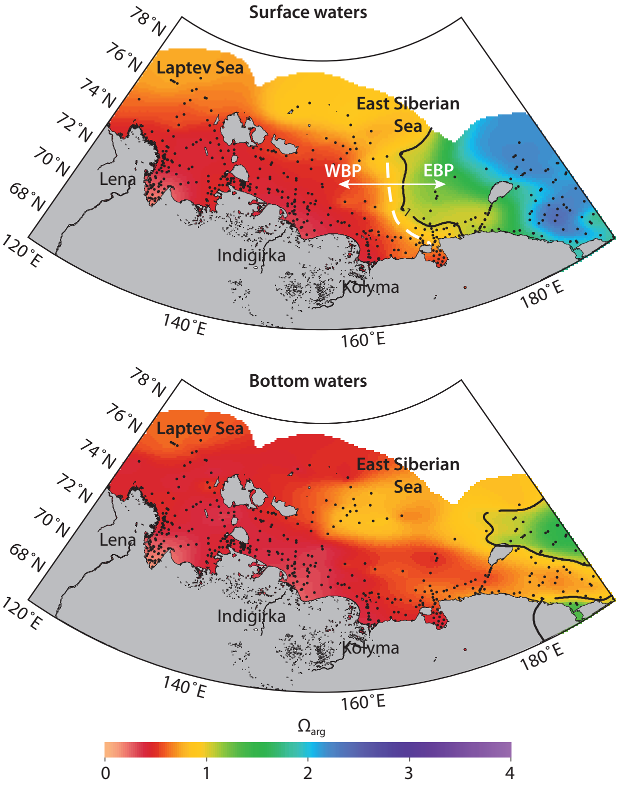


Figure 2.3 Distribution of Ω_{arg} over the East Siberian Arctic Shelf based on observations over the period 1999–2011 in surface waters and bottom waters. The white dashed curve shows the boundary between two biogeochemical provinces – the Western Biogeochemical Province and the Eastern Biogeochemical Province (modified from Semiletov et al., 2016).

Surface water Ω on the Western Biogeochemical Province reflects the freshening effect of river discharge, which also adds water with river-transported CO_2 and translocated terrigenous organic carbon, some of which gets degraded to CO_2 in the recipient shelf water, resulting in significantly decreased Ω_{arg} and over 3.5-fold lower saturations in the Western Biogeochemical Province than in the Eastern Biogeochemical Province (0.45 versus 1.74).

The low Ω observed in the bottom water in both biogeochemical provinces is determined by *in situ* organic carbon decomposition. Because rates of coastal erosion and the acidifying effect of terrestrial organic carbon input due to coastal erosion and river input are higher in the Western Biogeochemical Province, this results in mean Ω_{arg} in the Western Biogeochemical Province that is half as much as in the Eastern Biogeochemical Province (0.44 versus 0.88) (Semiletov et al., 2016). It was shown that in contrast to other marine ecosystems, where organic carbon originates from planktonic and riverine sources, coastal erosion represents a significant source of allochthonous organic carbon to the East Siberian Arctic Shelf (Fritz et al., 2017). Thus, Semiletov et al. (2016) have identified the important role of terrestrial organic carbon degradation in ocean acidification following permafrost loss along the East Siberian Shelf. It was found that the persistent acidification of East Siberian Arctic Shelf waters is driven by the degradation of terrestrial organic matter and discharge of Arctic river water with elevated CO_2 concentrations, rather than by uptake of atmospheric CO_2.

Terrestrial organic matter released from different sources and transported in different physical forms has different propensity towards both long-range transport and degradation (Vonk et al., 2014; Karlsson et al., 2016). The effect of terrestrial organic carbon on atmospheric CO_2 concentrations may not be a simple function of the terrestrial organic carbon flux to the ocean, but depends largely on its further fate upon coastal delivery (Bröder et al., 2018). Recent investigations provide a broad view on different terrigenous organic carbon pools and their behavior during cross-shelf transport (Vonk et al., 2014, 2017; Bröder et al., 2016a,b, 2018; Karlsson et al., 2016; Tesi et al., 2016; Brüchert et al., 2018). The terrestrial organic carbon released in the dissolved and colloidal forms is probably more exposed to degradation processes in the water column than sedimentary and particulate organic carbon (Karlsson et al., 2016). The studies point to a significant sink of terrestrial dissolved organic carbon (tDOC) over the broad continental shelves of the eastern Arctic, supporting the view that Arctic tDOC is partly biolabile and that the most labile fraction is delivered after the spring freshet (Alling et al., 2010; Letscher et al., 2011; Karlsson et al., 2016; Kaiser et al., 2017). *In situ* decay constants of tDOC vary in the range 0.18–0.58 per year, and the efficient removal of tDOC confirms that its mineralization has a strong impact on air-sea CO_2 exchange, and thus acidification in the Siberian Shelf seas (Kaiser et al., 2017). As the amount of river discharge continues to increase, along with increasing dissolved organic carbon export due to climate warming and thawing permafrost, the remineralization of terrigenous organic matter over the Arctic shelves should reduce the Arctic Ocean's ability to absorb atmospheric CO_2 and should increase the acidification of the seawater. The degree of degradation of sedimentary permafrost-derived terrestrial organic carbon is a function of the time spent under oxic conditions during protracted cross-shelf transport (Bröder et al., 2016a,b, 2018; Brüchert et al., 2018). For the wide East Siberian Arctic Shelf, long-lasting sediment transport allows for terrestrial organic carbon degradation, thereby constituting a carbon source to overlying water and atmosphere (Bröder et al., 2018). These results corroborate and expand previous findings for the East Siberian Arctic Shelf, showing that the shallowest shelf seas in this region function as an active reactor for terrigenous organic carbon in contrast to the Mackenzie Basin, which is thought to act as a geological sink for organic carbon due to its terrestrial organic carbon burial (Hilton et al., 2015; Bröder et al., 2016a). With ongoing global warming, rising permafrost-derived organic carbon input from river-sediment discharge and coastal erosion is expected to reach the marine environment.

In winter, the East Siberian Arctic Shelf water receives additional CO_2 from respired organic matter, especially in the bottom layer, where pCO_2 values increased up to 5000 µatm; the direct consequence is enhanced biologically driven acidification (Semiletov et al., 2013). Contemporary changes in the Arctic climatology are impacting on the carbon cycle and influencing the air-sea CO_2 flux. Larger areas of open water, longer ice-free periods, growing river discharge and the degradation of permafrost can increase the ability of the East Siberian Arctic seas to act as a CO_2 source (Semiletov et al., 2016; Pipko et al., 2017).

Of growing concern is the potential for increased methane (CH_4) seepage following ocean warming: this may induce rapid and massive ocean acidification (Bellerby, 2017). The East Siberian Arctic Shelf is a strong source of CH_4 to the atmosphere year-round – the conservative estimate of total annual CH_4 emissions to the atmosphere is about 17 Tg/y (Shakhova et al., 2014). The East Siberian Arctic Shelf contains the largest and arguably most vulnerable stores of subsea CH_4 and the state of subsea permafrost is becoming a key factor controlling CH_4 fluxes from the seabed to the water column (Shakhova et al., 2015). Current destabilization of subsea permafrost results in increasing permeability for gaseous CH_4 long preserved in seabed deposits within and beneath permafrost (Shakhova et al., 2017). Most of the CH_4 dissolves in the water column, building up an aqueous CH_4 inventory, and although CH_4 oxidation rates are very low (Shakhova et al., 2015), this could add to shelf water acidification. There remains substantial uncertainty regarding the CH_4-induced seawater acidification on the shallow East Siberian Arctic Shelf. Persistent and potentially increasing Ω_{arg} undersaturation of East Siberian Arctic Shelf water has already far exceeded projected levels for the 2100, which are based only on atmospheric CO_2 uptake.

2.2.2 Alaska, Bering, and Chukchi sectors

Several comprehensive data synthesis products have been published since the previous AMAP assessment on ocean acidification (AMAP, 2013), detailing the state of ocean acidification in Alaskan coastal waters and the Pacific Arctic Region (Bates, 2015; Mathis et al., 2015; Qi et al., 2017; Cross et al., 2018). Trends have emerged that clearly indicate the rapid progression of ocean acidification within these regions, including continued rapid surface ocean CO_2 uptake from the atmosphere and increasing carbonate mineral undersaturation (e.g., Evans et al., 2015; Cross et al., 2018). Continuous observations from moored sensors show Ω_{arg} values of 0.5 to emerge as early as July and to persist in the Bering Sea for at least five months (Mathis et al., 2012). Similar observations show that parts of the Beaufort Sea are undersaturated with respect to aragonite for nearly 80% of the year (Cross et al., 2018). These persistent and severe undersaturations have been shown to lead to carbonate mineral dissolution in some hotspots (Cross et al., 2013).

Vulnerability of Alaskan seas to ocean acidification results from the synergistic combination of natural carbon accumulation factors alongside the gradual build-up of anthropogenic CO_2. Overall, these waters are estimated to have absorbed ~66.5 µmol/kg anthropogenic CO_2 (Cross et al., 2013, 2018). Increasing Ω_{arg} undersaturation in the Bering, Chukchi and Beaufort Seas is a direct result of anthropogenic CO_2 (Mathis et al., 2011; Cross et al., 2013, 2018) despite the strong natural biogeochemical forcing from the biological pump, terrestrial carbon inputs, sea-ice formation and melt, and natural circulation processes (e.g., Cross et al., 2013, 2014; Yamamoto-Kawai et al., 2013, 2016; Evans et al., 2015; Strong et al., 2016).

Other mechanisms of climate change impacting high latitudes may have implications for ocean acidification. For example, some studies have hypothesized that primary production may increase due to changing light availability, stratification, and nutrient supply, thereby facilitating greater uptake of CO_2 over the Bering and Chukchi shelves (e.g., Tremblay et al., 2012) and remote sensing studies suggest these increases may be visible (e.g., Arrigo and van Dijken, 2015). It has also been hypothesized that this enhancement of the biological pump will in turn deliver more CO_2 to bottom waters, and more rapidly facilitate the development of Ω_{arg} undersaturation.

2.2.3 Central Arctic Ocean

Model simulations indicate that acidification of the central Arctic Ocean will increase with decreasing sea-ice cover, caused by increased CO_2 uptake and freshwater input. These results are largely supported by the CMIP5 assessment (Steiner et al., 2014). Popova et al. (2014) showed the spatial heterogeneity of ocean acidification in the Arctic, where the impact of climate change on the Arctic Ocean was more important than the increase in atmospheric CO_2. Applying an alternative model, Luo et al. (2016) highlighted the importance of inflowing water from the Atlantic Ocean on future desaturation of subsurface waters; a feature that has also been stressed based on observations (Ericson et al., 2014). The importance of freshwater for a number of processes, including its impact on biogeochemistry was summarized by Carmack et al. (2016), who also addressed the dependence of the freshwater distribution on the atmospheric pressure field.

2.2.4 Canada Basin and Beaufort Shelf

Long time series are already showing that undersaturated water masses have expanded both vertically (deeper) and horizontally (northerly) into the higher latitudes, both as a result of anthropogenic CO_2 and long-term temperature-driven changes in circulation that export more pre-conditioned water from the Chukchi shelf. For example, Qi et al. (2017) showed that, within Canada Basin, waters undersaturated with respect to aragonite expanded northward by at least 5° and deepened by ~100 m between 1994 and 2010. The undersaturated waters have expanded from 5% to 31% north of 70°N. Qi et al. (2017) indicated that overall, the percentage of aragonite undersaturated vertical area of the water column (0–250 m and between 70° and 90°N) increased six-fold from 5% in 1994 to 31% in 2010, with an average rate of increase of 1.5% per year Extrapolation of this rate suggests that the entire upper water column (0–250 m) in the western Arctic Ocean will be undersaturated with respect to aragonite by 2055.

Robbins et al. (2013) concluded that ~20% of the Canada Basin and Makarov Basin surface waters were undersaturated with respect to aragonite, an area with substantial sea ice melt. Wynn et al. (2016) showed that a second undersaturated zone at ~90–220 m depth and salinity ~31.8–35.4 occurs within the Arctic halocline and is characterized by elevated pCO_2 and elevated nutrient concentrations. The nature of this horizon is consistent with remineralization of organic matter on shallow continental shelves bordering Canada Basin and the input of nutrients and CO_2 entrained by currents from the Pacific Inlet. Anderson et al. (2017) described the export of the undersaturated water from the East Siberian Sea and traced its path through large parts of the central Arctic Ocean (see also Annex 1).

The Beaufort Shelf is one of the shelves especially vulnerable to ocean acidification, since it experiences naturally undersaturated Pacific seawater inflows created upstream by high primary productivity and resultant organic matter decay in the Bering and Chukchi Seas combined with the generally high CO_2 content of old, deep Pacific upwelling water. Local and seasonal effects on the shelves are also highly influential. On the Beaufort Shelf, cations are diluted by inflows from the Mackenzie River and from sea-ice melt. High productivity over the shelves in summer, terrestrial organic carbon supplies from rivers and rapid coastal erosion, and localized upwelling ultimately result in further reduction of pH and $CaCO_3$ saturation state (AMAP, 2013). Tynan et al. (2016) indicated large gradients in surface pH and Ω_{arg} over small spatial scales, which were particularly strong in sea-ice covered areas; up to 0.45 in pH and 2.0 in Ω_{arg}. Such gradients frequently arise due to spatial variability in primary production and remineralization processes. Additional reduction in pH occurs under ice due to the release of CO_2-enriched brines during ice formation. Miller et al. (2014) evaluated changes in the marine carbonate system of the western Arctic and found substantial changes since the 1970s. Spatially averaging observations across the Beaufort Sea and Canada Basin, they found that the mean at the surface was still supersaturated, i.e., $\Omega_{arg} > 1$, but that upper halocline waters and deep waters regularly showed aragonite undersaturation towards the end of the study period. Figure 2.4 shows updates from Miller et al. (2014), where data have been separated for on and off shelf areas (i.e. shallower and deeper than 250 m, respectively). Off-shelf waters show even stronger reductions in saturation state in the upper 200 m of Canada Basin than indicated by Miller et al. (2014), while the Beaufort Shelf waters still remain supersaturated. Mol et al. (2018) also confirm that during the autumn-summer period surface waters of the Mackenzie Shelf were supersaturated with respect to aragonite. Undersaturated waters were found in the shelf bottom layer and in the Pacific-origin upper halocline layer. These waters, centered at a salinity of 33.1 and a temperature minimum, have pCO_2 values over 600 µatm and Ω_{arg} as low as 0.83. Mol et al. (2018) observed that aragonite undersaturated waters with high pCO_2 were restricted to depths of more than 20 m over the shelf at the time of sampling, but concluded that a strong upwelling wind event could easily bring this water to the surface. Tracer data and model simulations suggest that increased Pacific Winter Water transport, driven by an anomalous circulation pattern and sea-ice retreat, is primarily responsible for the expansion, although local carbon remineralization and increased CO_2 uptake due to increases in atmospheric CO_2 concentrations have also contributed. In summary, the assessment confirms that more rapid acidification is occurring in the Arctic Ocean than the Pacific and Atlantic oceans, with the western Arctic Ocean being the first open-ocean region with large-scale expansion of undersaturated waters directly observed in the upper water column. Measurable downstream effects on seawater chemistry are propagated via M'Clure Strait and Amundsen Gulf into waters over the Canadian Polar Shelf and beyond.

2.2.5 Canadian Polar Shelf, Baffin Bay, Davis Strait and the West Greenland regions

A study from the North Pacific via Canada Basin and the Canadian Polar Shelf to the North Atlantic showed the evolution of saturation states with controlling mechanisms along the flow pathways (Yamamoto-Kawai et al., 2013). The preconditioned Pacific Water with low Ω due to high concentrations of remineralized carbon from global circulation history, enters into the Arctic. The Ω is further lowered following the addition of freshwater from rivers and sea-ice melt. Consequently, the Arctic outflow over the Canadian Polar Shelf has low Ω, and a strong Ω front between Arctic and Atlantic waters forms above Davis Strait. The study of Yamamoto-Kawai et al. (2013) confirmed the findings of an earlier study demonstrating the influence of

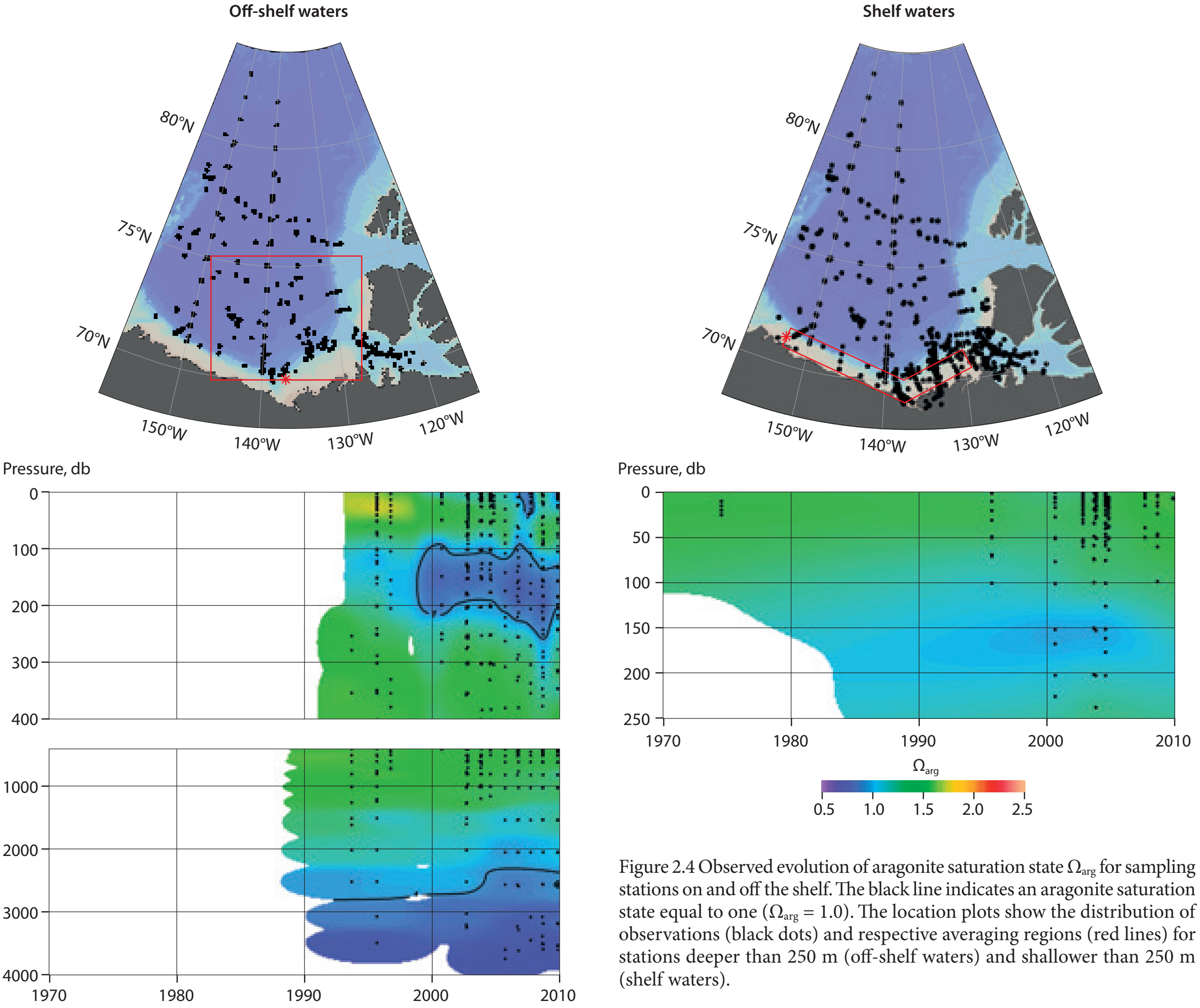


Figure 2.4 Observed evolution of aragonite saturation state Ω_{arg} for sampling stations on and off the shelf. The black line indicates an aragonite saturation state equal to one ($\Omega_{arg} = 1.0$). The location plots show the distribution of observations (black dots) and respective averaging regions (red lines) for stations deeper than 250 m (off-shelf waters) and shallower than 250 m (shelf waters).

the Arctic outflow on saturation states and pH in the Canadian Polar Shelf, Baffin Bay/Davis Strait and the Labrador Sea (Azetsu-Scott et al., 2010). In a west Greenland fjord, the surface layer of the entire fjord and adjacent continental shelf are undersaturated in CO_2 throughout the year and glacial meltwater is an important driver for this undersaturation of CO_2 (Meire et al., 2015). This CO_2 undersaturation can be explained by the thermodynamic effect on the surface water pCO_2 resulting from the mixing of fresh glacial meltwater and ambient saline fjord water. In a Baffin Island fjord, higher contribution of meteoric water (glacier meltwater and river runoff in this study) in surface waters resulted in lowering total alkalinity, dissolved inorganic carbon and calcium ion and, thus, lowering Ω compared to the surrounding Baffin-Labrador region (Turk et al., 2016). In contrast, a study of kelp forests in Nuuk and Disko Bay in West Greenland showed that long photoperiods in Arctic summers sustained up-regulation of pH, which can have potential benefits for calcifying organisms (Krause-Jensen et al., 2015, 2016; Krause-Jensen and Duarte, 2016). The Hudson Bay system was shown to be strongly influenced by river runoff. Aragonite undersaturation was observed in the surface water of south-eastern Hudson Bay, where the river runoff fraction was high (>10%). The watershed geology (e.g., limestone vs. igneous rocks) influenced the alkalinity of river runoff in different parts of Hudson Bay, which contributed to Ω variation in the coastal region (Azetsu-Scott et al., 2014). The timing and magnitude of freshwater inputs also play an important role in carbon biogeochemistry and acidification in this unique Arctic ecosystem (Burt et al., 2016). The distributions of carbonate parameters throughout the Hudson Bay system in July 2010 were similar to those reported by Azetsu-Scott et al. (2014) for September 2005. Ω_{arg} in deep waters shows no discernable change from 2005 levels, and slight changes in surface water Ω_{arg} can be attributed to seasonal variability (Burt et al., 2016). Model studies indicate that the impact of changing temperature, stratification, primary production and freshwater fluxes play a strong role in the declines in pH and Ω seen in the Arctic (Steiner et al., 2015). The central Arctic, Canadian Polar Shelf and Baffin Bay show the greatest rates of acidification and pH decline as a result of melting sea ice.

2.2.6 Greenland, Iceland and Norwegian seas

Flowing out of the Arctic Ocean between Greenland and Svalbard, the surface and intermediate waters mix into the Greenland and Iceland seas transporting carbon into the intermediate and deep waters. The Greenland Sea exhibits significant ocean acidification, with reductions in pH of 0.0023 units per year in the surface 200 m between 1981 and 2013 (Skjelvan et al., 2014; Figure 2.5). Significant penetration of anthropogenic carbon deep into the water column to

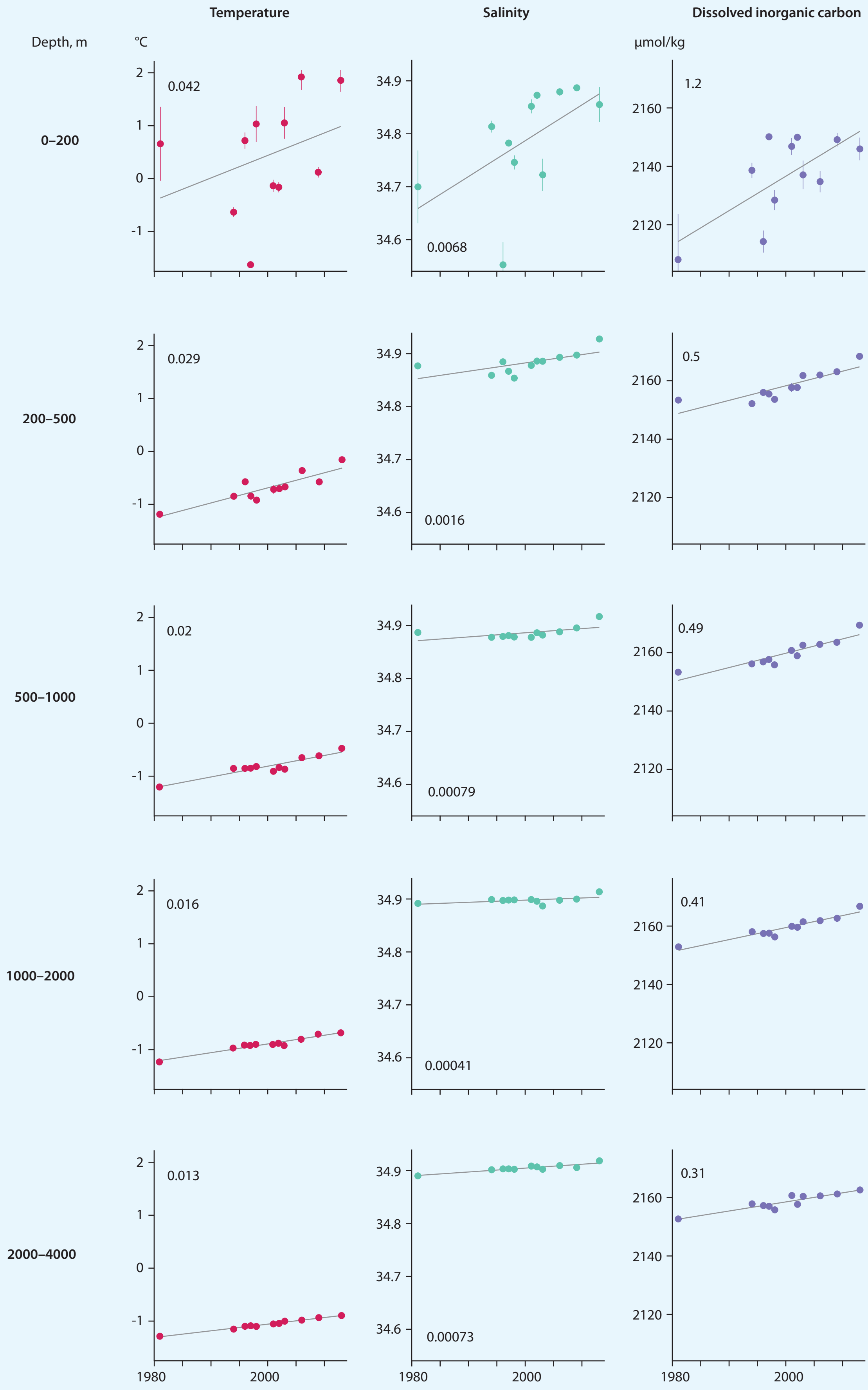
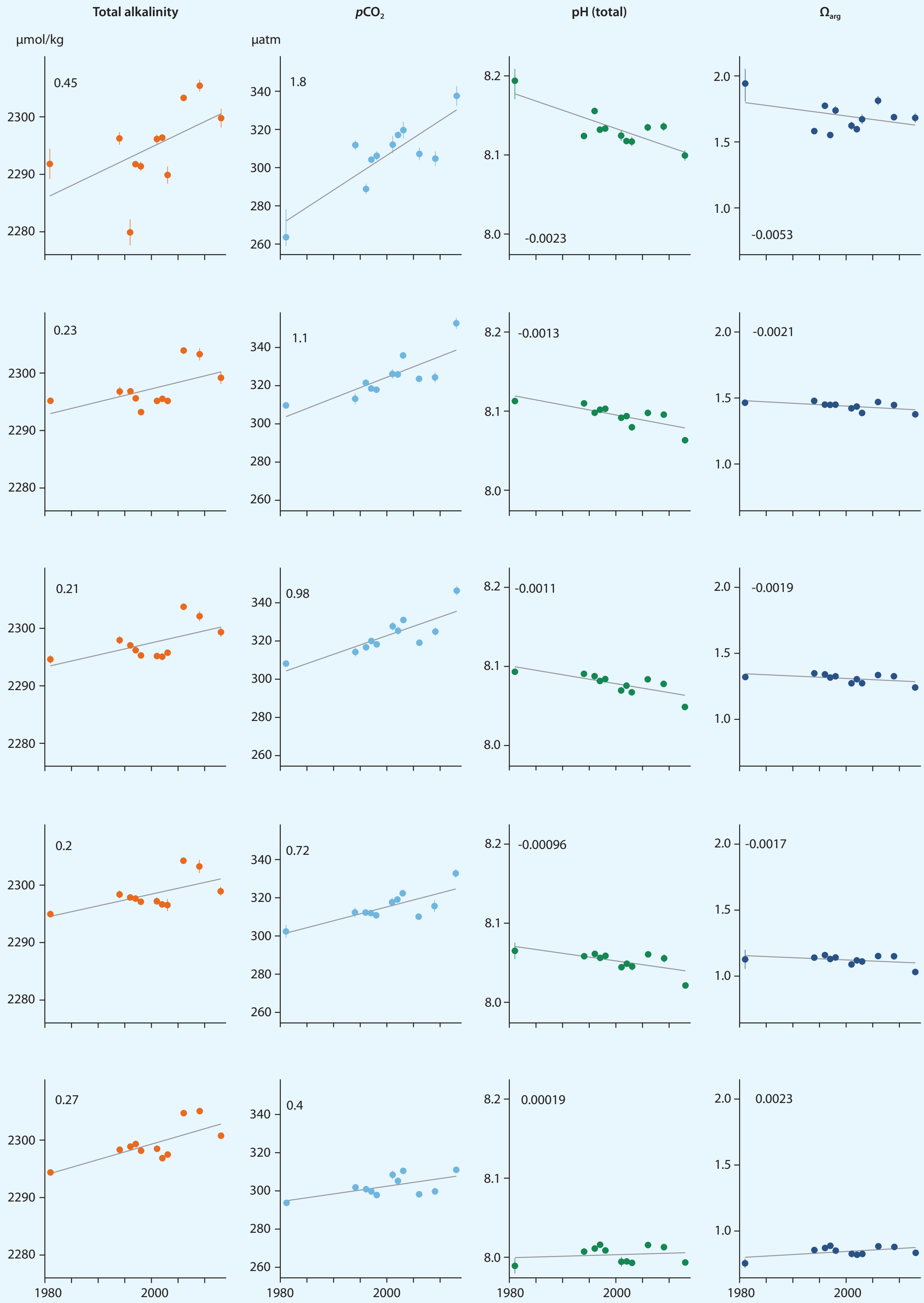


Figure 2.5 Trends in hydrography and the marine carbonate system for the Greenland Sea (based on Skjelvan et al., 2014).


Total alkalinity
pCO_2
pH (total)
Ω_{arg}
µmol/kg
µatm
0.45
1.8
-0.0023
-0.0053
0.23
1.1
-0.0013
-0.0021
0.21
0.98
-0.0011
-0.0019
0.2
0.72
-0.00096
-0.0017
0.27
0.4
0.00019
0.0023
2280
2290
2300
260
280
300
320
340
8.0
8.1
8.2
1.0
1.5
2.0
1980
2000

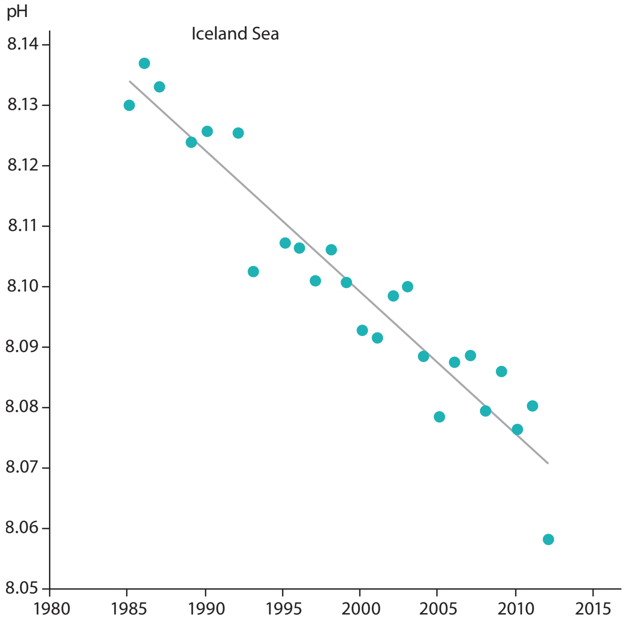


Figure 2.6 Iceland Sea surface water pH_T (total pH) changes in winter (based on Olafsson et al., 2009).

2000 m was also documented. The Iceland Sea in recent years has shown gradual trends towards warmer and saltier surface waters (Larsen et al., 2016). The East Greenland Current covers the westernmost part of the Iceland Sea. The southern extent of Arctic sea ice in the East Greenland Current has retreated northward making large areas of Polar Water ice free in summer. Time series observations conducted in the central gyre of the Iceland Sea have previously revealed a high rate of surface water acidification (Figure 2.6). The acidification reaches into the Arctic Deep Water where the calcium carbonate state of undersaturation, $\Omega_{arg} <1$, is at about 1700 m depth and is shoaling at a rate of about 4 m/y spreading undersaturation over new sea floor areas (Olafsson et al., 2009). Seasonality of the surface carbonate system in the Iceland Sea is strong, however, results from surface mooring observations recording surface carbon chemistry suggest that the present-day pH and Ω_{arg} conditions now fall outside the bounds of pre-industrial variability (Sutton et al., 2016). A major contribution to the acidification rate observed for the Iceland Sea is air-sea flux of atmospheric CO_2. An evaluation based on observations of carbon and nutrients from the Iceland Sea time series station indicated that the annual air-sea uptake of atmospheric CO_2 amounted to 4.4±1.1 mol C/m^2/y (Jeansson et al., 2015).

2.3 Projections of Arctic ocean acidification

Projections of future conditions performed with earth system models (ESMs) consistently show enhanced ocean acidification in polar regions (e.g., Steinacher et al., 2009; Denman et al., 2011; Joos et al., 2011; AMAP, 2013; Steiner et al., 2014) and suggest that Arctic Ocean acidification will continue over the next century with accelerated reductions in Ω at least until the sea-ice cover reaches a new steady state with largely ice-free summers (Steiner et al., 2014, Figure 2.7).

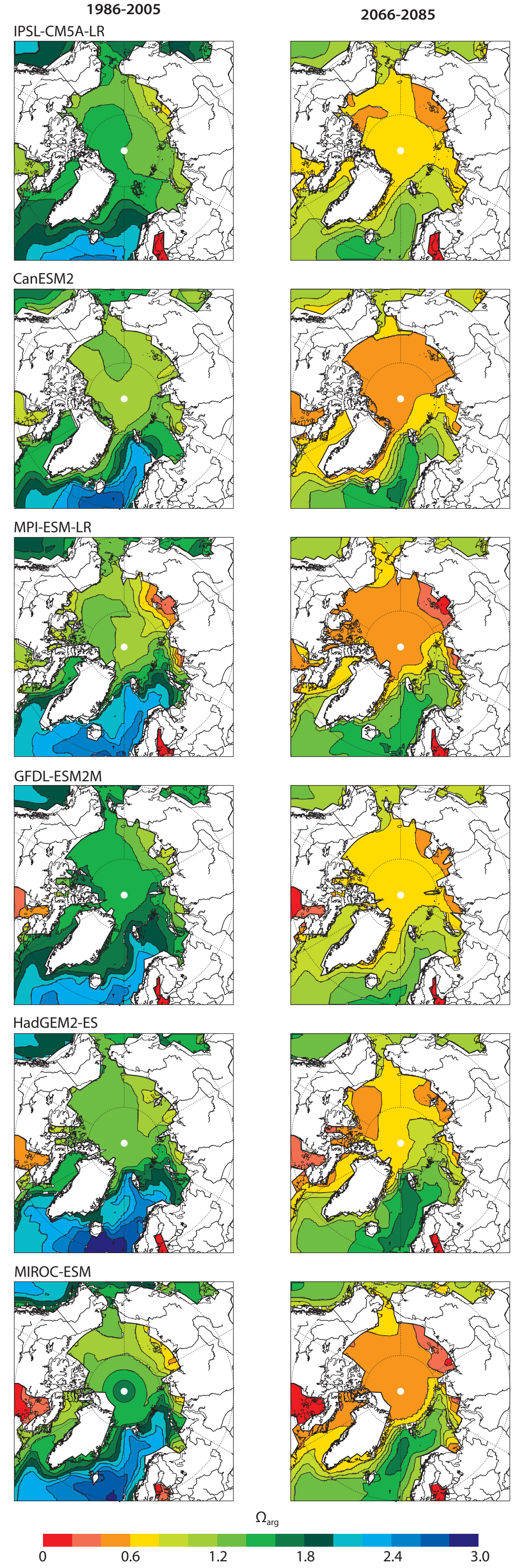


Figure 2.7 Surface aragonite saturation state Ω_{arg} as simulated under the RCP8.5 scenario by six ESMs (Steiner et al., 2014).

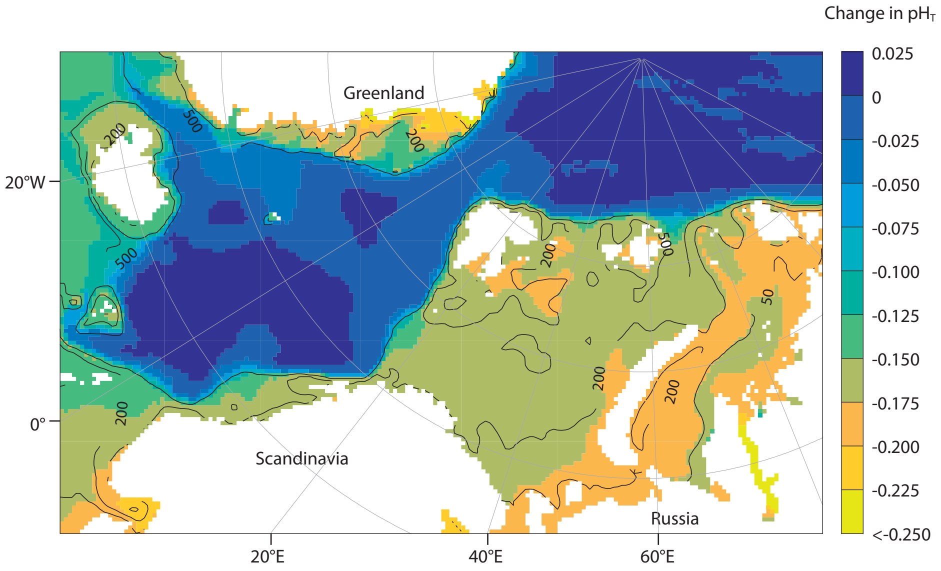


Figure 2.8 Reduction in bottom water pH_T (total pH) between bi-decadal averages 2000–2019 and 2050–2069 (Wallhead et al., 2017).

Projected increased freshwater addition due to melting of glaciers as the result of warming will further decrease $CaCO_3$ saturation and increase ocean acidification (Piquet et al., 2014). Winter-surface and bottom water Ω may further decrease due to increased accumulated freshwater from summer ice melt. Previous studies show that Atlantic Water contains high amounts of anthropogenic CO_2 (e.g., Sabine et al., 2004) due to efficient CO_2 uptake during cooling as the Atlantic Water is transported northward along the Norwegian coast. Olsen et al. (2006) estimated the rate of fCO_2 increase due to anthropogenic CO_2 uptake to be about 1 µatm/y in the West Spitzbergen Current and on the West Spitsbergen shelf, which will increase ocean acidification. A model study by Skogen et al. (2014) predicted a doubling of the ocean CO_2 uptake during the 2000–2065 period from 1.9 to 3.1 mol C/m^2/y in the West Spitzbergen Current and Atlantic Water, resulting in a decrease in pH of -0.19 over this period. Fransson et al. (2016) assumed two scenarios in Svalbard fjords: a colder and less saline Arctic regime and a warmer and more saline Atlantic regime. For example, in Kongsfjorden, the Transformed Atlantic Water is influenced by the inflow of either a greater component of colder and less saline and lower Ω water transported by the coastal current, or by the inflow of warmer and more saline and higher Ω water.

Model estimates of ocean acidification suggest that the Barents Sea, and the waters surrounding Svalbard will see a large pH decrease – about 0.25 to 0.35 by 2065 (Skogen et al., 2014). The sub-surface (50–500 m) waters of the Norwegian, Barents, Kara, and East Greenland shelves are projected to undergo a rapid and spatially variable decline in bottom pH of 0.10–0.20 units over the next 50 years (Wallhead et al., 2017; Figure 2.8). Bottom water undersaturation with respect to aragonite occurred over the entire Kara shelf by 2040 and over most of the Barents and East Greenland shelves by 2070 (Wallhead et al., 2017).

The projected onset of Ω_{arg} undersaturation is highly variable regionally within the Arctic, occurring during 2000–2010 on the Siberian shelves and in the Canadian Arctic Archipelago, and as late as the 2080s in the Barents and Norwegian seas (Popova et al., 2014).

Projections based on global models indicate that surface waters of the Bering, Chukchi and Beaufort seas corroborate this result, suggesting that the annual average Ω_{arg} values will be corrosive between 2027 and 2052 under a business-as-usual emissions (RCP8.5) scenario (Mathis et al., 2015). Large natural variability in the Bering Sea is likely to make this region more resilient to declining pH, however, by 2085 the Bering Sea is projected to exceed the range of its natural carbonate system variability resulting in tremendous pressures on commercial and subsistence fisheries. To better resolve issues of fine-scale spatial and temporal variability, biogeochemistry is now being incorporated into some regional models. Multi-model comparison of bi-decadally averaged annual mean surface pH and Ω_{arg} in Baffin Bay show a pH decrease from 8.16 in 1966–1985 to 7.80 in 2066–2085 and a Ω_{arg} decrease from 1.52 to 0.74 during the same period, with aragonite undersaturation expected around mid-century (Steiner et al., 2014). Projections with RCP8.5 forcing for Canada Basin showed reductions in the bi-decadal mean surface pH from about 8.1 in 1986–2005 to 7.7 by 2066–2085, closely linked to reductions in Ω, from about 1.4 (2.0) to 0.7 (1.0) for aragonite (calcite). Under RCP4.5 (a scenario with emission mitigation) the progression of ocean acidification is reduced and pH of 7.9 is reached about 25 years later than for RCP8.5. Nevertheless, the emergence of undersaturated surface waters, which is projected to occur within the next decade, differs little between the two scenarios (Steiner et al., 2014). While ESMs consistently project continued progression of ocean acidification in the Arctic, their horizontal resolution is mostly insufficient to allow focus on regional details, especially along coastal and shelf areas. Projections performed with higher resolution ocean-only models provide more detailed projections for the region, while showing very similar trends and pan-Arctic patterns. For the central Canadian Polar Shelf, the projections show a continuous decline across all depths over time for both Ω_{arg} and Ω_{cal}. By 2066–2085 all of the central Canadian Polar Shelf is projected to be undersaturated with respect to Ω_{arg}, and bottom undersaturation will be reached for Ω_{cal}. The projected enhanced acidification happens in parallel to a continuous increase in temperature across all depths and significant changes in sea ice cover, affecting mixing, and light penetration.

2.4 Conclusions and recommendations

Sustained observations of the Arctic Ocean witness a rapidly changing carbonate system and ensuing ocean acidification. The modifications to the Arctic Ocean carbon cycle portray a very heterogeneous system with large variations, seasonally and regionally, in the foundations for this ocean acidification. An increasing source is from the atmosphere, but also oceanic advection of excess carbon from outside the Arctic Ocean. On regional to local scales, ocean acidification is also accelerated through changes to the freshwater cycle and from the injection of organic carbon, to be later respired, from terrestrial sources. Increasing ice melt is also influencing the marine carbonate system. Nevertheless, understanding of the seasonality, trends and future projections is limited due to a lack of consolidated measurements at pan-Arctic dimensions and to cover the seasonal scale, with most observations made in ice-free months. There is an urgent need to further develop technological solutions to monitor ocean acidification at high frequency, from autonomous sensors, year-round, and over the entire water column. There is a lack of international coordination and long time-series frameworks; observations are generally ad hoc and reliant on successful project funding and thus are not of the scale to incorporate the wide range of disciplinary science required to understand the rates, causes and impacts of ocean acidification. As a result, ocean acidification observations are not always tailored to deliver the temporal and spatial information relevant for assessing sensitivity of key Arctic marine ecosystems and their services. Ongoing coordination efforts through the Global Ocean Acidification Observing Network (GAO-ON) are underway to deliver new knowledge on ocean acidification towards the UN Sustainable Development Goals associated with Agenda 2030. The joint IMBeR / Future Earth Coasts Continental Margins working group, has the goal of developing the socio-ecological understanding of ecosystem multi-stressors in the Arctic marginal seas, including targeted ocean acidification observations, at an internationally coordinated level. At present, a bottom-up, researcher-driven initiative 'Synoptic Arctic Survey' is underway that seeks to define the present state of the Arctic Ocean and provide understanding of the major ongoing transformations, with an emphasis on water masses, the marine ecosystems, and the carbon cycle through an international ship-based field campaign in 2020.

3. Biological responses to ocean acidification

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3.1 Introduction

This chapter presents an overview of the current state of understanding regarding potential responses of Arctic species and ecosystems to ocean acidification. The focus of this qualitative literature review is on research conducted within the Arctic region that has been published within the past five years (i.e., since the first AMAP assessment on ocean acidification; see AMAP, 2013). However, for some ecosystem components there is still little information and so these sections have been supplemented by earlier studies and work from other geographical regions (details summarized in the Appendix to this chapter). The key taxonomic groups (viruses, bacteria and archaea, phytoplankton, foraminifera, macroalgae, corals, mollusks, echinoderms, crustaceans, other invertebrates, fishes, and seabirds and mammals), and communities they form, are considered in terms of their responses to ocean acidification, with potential implications for ecosystem-wide and longer-term acclimation and adaptation responses highlighted. It is also identified how ocean acidification may be modified by human activities driving changes in other abiotic characteristics such as temperature, light, nutrient availability, and the biota present. Together, the research reviewed in this chapter highlights the potential complexity of the biological effects of ocean acidification. Despite these varied responses, it does appear that ocean acidification is likely to be sufficient to drive changes in Arctic organisms and ecosystems to an extent that will impact the associated human societies.

3.2 Responses of key organisms

3.2.1 Viruses

Ocean acidification has the potential to affect viruses both directly via effects on the organism, and indirectly via effects on the host or species affecting the host. In the context of direct effects, the impact of ocean acidification has been found to be limited for many aquatic viruses; marine virus isolates have been quite stable in terms of particle decay and loss of infectivity over the range of pH associated with near-future ocean acidification (Crawfurd et al., 2017). Moreover, ocean acidification has had no direct effect on viral replication cycles (Tsiola et al., 2017), growth cycles (Maat et al., 2014), abundances (Rochelle-Newall et al., 2004; Celussi et al., 2017), lysis (Carreira et al., 2013; Maat et al., 2014), or burst size (a term describing lytic infections, number of phages per cell; Carreira et al., 2013). It is worth noting, however, that there have been some instances of ocean acidification affecting viruses. The diverse responses could reflect differences between the organisms or populations considered in the experimental protocols used, or other environmental conditions modified in combination with ocean acidification (discussed further in Section 3.5). An example of a virus responding to acidification was identified where burst size decreased with decreasing pH (although this was likely to be an indirect result of reduced host growth rate with decreasing pH; Traving et al., 2013). In contrast, increasing burst size has also been linked with decreasing pH (along with slightly delayed lysis) (Carreira et al., 2013). In addition, there has been a report of reduced viral abundances under elevated partial pressure of carbon dioxide ($p\mathrm{CO_2}$) (350 *v* 700 *v* 1050 μatm) in a mesocosm experiment conducted in Raunefjorden outside Bergen, Norway (Larsen et al., 2008).

Ocean acidification may have indirect effects by modifying interactions between viruses and the organisms they infect. In non-Arctic environments, decreasing pH has been found to influence the infection capacity of a cyanobacterial virus and its infection cycle (e.g., eclipse period, latent period), potentially modifying the effect of viruses on their hosts (Traving et al., 2013; Chen et al., 2015). Moreover, where viral population sizes or abundances are modified (either due to direct effects on the viruses, or indirect effects mediated by the hosts) (e.g., Larsen et al., 2008), this could affect Arctic ecosystems by modifying the marine pelagic food web.

3.2.2 Bacteria and archaea

Many of the studies highlighting potential responses of Arctic bacteria to ocean acidification come from the EPOCA (European Project on Ocean Acidification) 2010 Arctic campaign. This campaign included a large-scale *in situ* pelagic mesocosm study of nine experimental units within which a complex array of organisms were held and a range of $p\mathrm{CO_2}$ treatments were applied. The mesocosms were deployed in Kongsfjorden whose waters are Atlantic dominated (detail in Riebesell et al., 2013). In terms of bacterial functioning, here increased $p\mathrm{CO_2}$ was found to have no effect on processes related to bacterial carbon metabolism such as respiration, carbon demand, and growth efficiency (Motegi et al., 2013). Experiments subsequently conducted during June 2012 in the Arctic, specifically in the Baltic Sea at Tvärminne Storfjärden on the southwestern coast of Finland, have indicated no consistent $p\mathrm{CO_2}$ effect on bacterial protein production, cell-specific bacterial protein production, or biovolumes of either free-living or particle-associated heterotrophic bacteria when considered as individual components in univariate analyses (Hornick et al., 2017). In the EPOCA experiments, increased $p\mathrm{CO_2}$ did, however, lead to an increase in the extracellular enzyme activity of β-glucosidase (which catalyzes hydrolysis of glycosidic bonds) and leucine-aminopeptidase (which catalyzes hydrolysis of leucine residues at the N-terminus of peptides and proteins) (Piontek et al., 2013). A later study revealed an optimal pH of below 6 for β-glucosidase and of between 6.7 and 7.6 for leucine-aminopeptidase. That these optima are below current seawater pH of the study region indicates that there may be modified hydrolytic activity within bacteria under future ocean acidification (Piontek et al., 2015).

Scientists sampling mesocosms during an outdoor experiment investigating the reactions of marine organisms to ocean acidification at Kongsfjord, Ny-Alesund, Svalbard.

Communities of pelagic bacteria have been considered in the context of ocean acidification. In the EPOCA experiments, the bacterial community attached to particles was found to be more diverse at higher pCO_2 conditions (Sperling et al., 2013). In contrast, where bacterioplankton characteristics such as diversity, taxonomic richness and community structure were considered in these experiments, they were found to be influenced principally by variation in primary production under ocean acidification rather than direct effects of pCO_2 (Zhang et al., 2013). An additional study recently conducted in the Arctic Ocean (Svalbard) supports this finding; the application of phylogenetic molecular ecological networks identified that elevated pCO_2 did not affect microbial community structure or succession (Wang et al., 2016) (but see Hornick et al., 2017 for results suggesting microbial community composition and complex trophic interactions may be altered under a future acidified ocean). Within the EPOCA results there was, however, a negative correlation between abundance of the phylum Bacteroidetes and pCO_2 by the end of the experiment (Zhang et al., 2013). Where lower bacterial abundances were identified at increased pCO_2, this was suggested to be an effect of higher rates of viral lysis (Brussaard et al., 2013). Analysis of rare bacterial taxa identified 15 taxa correlated with pCO_2 treatment and, in contrast to the result noted above, most of these increased in abundance with higher pCO_2 (i.e., a positive correlation) (Roy et al., 2013). Together, the results of these different studies indicate the potentially varied responses that may be observed in Arctic pelagic bacterial communities experiencing ocean acidification.

Arctic marine surface sediments contain active bacterial communities that can respond to surrounding environmental conditions. In terms of their diversity/similarity, it has been found that these communities may show very little difference between pCO_2 treatments – it was only when the highest and lowest treatments were considered (380 v 3000 µatm) that significant differences were evident. Specifically, there were increases in the abundances of operational taxonomic units most closely related to Halobacteria and differences in the presence/absence structure of the Planctomycetes. In addition, the relative abundance of members of the classes Planctomycetacia and Nitrospira increased with increasing pCO_2 concentration (Tait et al., 2013). A subsequent study showed similar patterns; following exposure of sediment cores to experimental pCO_2 conditions, increased pCO_2 led to an increase in the abundance of Planctomycete-specific 16S rRNA (the vast majority of which grouped with known ammamox bacteria). There was, however, no change to the abundance of bacterial *amoA* genes (which encode the active site of the enzyme ammonia monooxygenase), but the abundance of archaeal *amoA* transcripts was reduced accompanied by a shift in the composition of the active community which could have broader ecosystem-level consequences. These results suggest bacteria and archaea have different pH optima, potentially meaning that their activities and role in the nitrogen cycle may be differentially affected by ocean acidification (Tait et al., 2014).

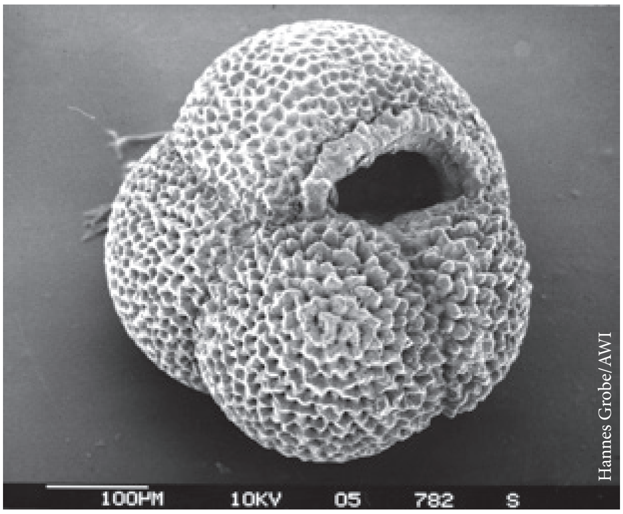
When the effects of ocean acidification on under-ice bacteria in the Arctic have been investigated results indicated that increased pCO_2 may have little effect on dominant taxa, although diversity may be reduced in some orders (Monier et al., 2014).

There have been relatively few studies considering the responses of archaea to ocean acidification, yet they represent 10–40% of picoplankton in the Arctic (Kirchman et al., 2007). The few studies that do exist indicate that archaea may be relatively unaffected by ocean acidification (Hassenrück et al., 2016; Currie et al., 2017). As noted above, however, the abundance of archaeal *amoA* transcripts may be reduced under ocean acidification, a change accompanied by a shift in the composition of the active community (Tait et al., 2014).

3.2.3 Phytoplankton

The effects of ocean acidification on phytoplankton have mostly been studied in single species laboratory experiments covering major functional groups including silicifying diatoms (Pancic et al., 2015; Heiden et al., 2016; Wolf et al., 2018), calcium carbonate-producing coccolithophores (Sett et al., 2014; Kottmeier et al., 2016), nitrogen-fixing cyanobacteria (Fu et al., 2007), and chlorophytes (Maat et al., 2014; García-Gómez et al., 2016). The focus of these experiments is typically on key physiological aspects such as carbon fixation, calcification, growth or nitrogen fixation rates, but also extends to trace gas production or modes of inorganic carbon acquisition (e.g., Fu et al., 2007; Sett et al., 2014; Pancic et al., 2015; García-Gómez et al., 2016; Heiden et al., 2016; Kottmeier et al., 2016; Webb et al., 2016; Wolf et al., 2018). Overall, Arctic and subarctic phytoplankton communities seem to be comparatively resilient to ocean acidification with no significant change in net primary production and little change in species composition under enriched scenarios up to 1000 µatm $p\mathrm{CO}_2$ (Hoppe et al., 2017, 2018). When effects of ocean acidification are found small picoeukaryotes seem to benefit (Davidson et al., 2016; Hussherr et al., 2017), whereas haptophytes (including coccolithophores) are typically negatively impacted (Yoshimura et al., 2013; Thoisen et al., 2015), and varying responses are observed in diatoms (Coello-Camba et al., 2014; Hoppe et al., 2017, 2018) and the cyanobacterium *Synechococcus* (e.g., Schulz et al., 2017 and references therein; Segovia et al., 2017). This general pattern was also reported in a recent review with global coverage and, given the multitude of experimental approaches and designs incorporated, appears to be a robust finding (Schulz et al., 2017).

Where species-specific responses to ocean acidification are observed, the composition of phytoplankton communities are expected to shift significantly, potentially modifying interactions with other ecosystem components (further details on species interactions can be found in Section 3.3). Interactions that can be modified include those associated with trophic links. That is, where phytoplankton biochemical composition is modified this may alter the quality of food through which energy is transferred to higher trophic levels. There have been reports that ocean acidification decreased polyunsaturated fatty acid (PUFA) concentrations in phytoplankton which resulted in a reduction of these essential components in copepods, negatively impacting somatic growth (e.g., Bermudez et al., 2016a) and egg production (e.g., Rossoll et al., 2012). However, no effect on phytoplankton (Bermudez et al., 2016b), and positive effects on mesozooplankton PUFA content have also been found (Wang et al., 2017), as have increases in zooplankton biomass; these changes were most likely to have been driven by ocean acidification induced changes in phytoplankton community structure (e.g., Taucher et al., 2017). As none of these experiments were conducted in high latitude waters, and given the range of responses (also compare, for example, Cripps et al., 2016 and Garzke et al., 2016), the impacts of ocean acidification on phytoplankton-zooplankton interactions in the Arctic remain poorly understood.



Hannes Grobe/AWI

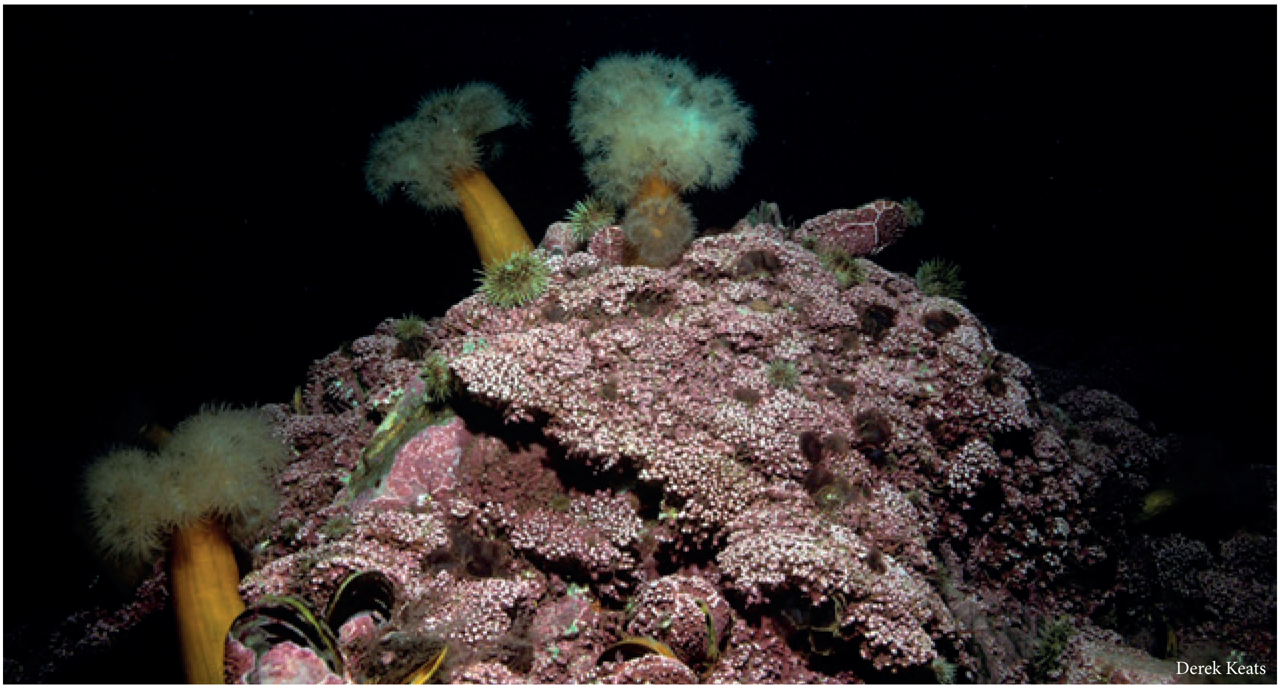
The planktonic foraminifera *Neogloboquadrina pachyderma.*

3.2.4 Foraminifera

Assemblages of planktonic foraminifera in the Arctic are typically dominated by the species *Neogloboquadrina pachyderma.* When experimentally exposed to increased $p\mathrm{CO}_2$, this species has been found to show no response in terms of survival, but individuals have smaller shell diameters (Manno et al., 2012). Several *Globigerina* species are also found in these Arctic phytoplankton assemblages. One of them, *G. bulloides*, was experimentally exposed to lower seawater pH which was associated with decreased shell calcification and repair of spines (along with oxygen consumption) (Davis et al., 2017).

Benthic foraminifera in a range of environments, from fjords to coral reefs, have shown diverse responses to ocean acidification. For example, in terms of survival, while some studies have found acidification is likely to have no effect (McIntyre-Wressnig et al., 2013, 2014), reduced survival has also been identified in another study (Marques et al., 2017). The reason for these differences can reflect either physiological differences or differences in experimental design (e.g., scale of experiments, season conducted, study location). Other processes considered for benthic foraminifera under ocean acidification include growth, which has sometimes been found to be decreased (Sinutok et al., 2011), but may also remain unchanged (McIntyre-Wressnig et al., 2013; Prazeres et al., 2015). Where changes are observed, these are likely to be influenced by modifications in other physiological processes, as can be indicated by increased net oxygen production (Uthicke and Fabricius, 2012), the reduced rate of oxygen production (Sinutok et al., 2011), increased respiration (Uthicke and Fabricius, 2012), reduced Ca^{2+}-ATPase activity (Marques et al., 2017), inhibition of Mg-ATPase (Prazeres et al., 2015), increases of both Ca-ATPase and Mg-ATPase activities (Prazeres et al., 2015), or declines in photosynthetic processes (Sinutok et al., 2011).

A range of organism characteristics has been considered in connection with the shells of benthic foraminifera under ocean acidification. Calcification has been found to be reduced (Fujita et al., 2011; Sinutok et al., 2011; Uthicke and Fabricius, 2012; Reymond et al., 2013), remain unchanged (Vogel and Uthicke, 2012), and even increase (Vogel and Uthicke, 2012) where $p\mathrm{CO}_2$ is manipulated. The shells that exist under ocean



Thick coralline crust (mainly *Lithothamnion glaciale*) with anemones (*Metridium senile*), Newfoundland.

acidification may have reduced weight (Dissard et al., 2010), smaller diameter (Haynert and Schönfeld, 2014; Marques et al., 2017), modified chemical composition (Dissard et al., 2010; Robbins et al., 2017), reduced density of inner chambers (Prazeres et al., 2015) (but see Prazeres et al., 2015 for an example of no change in a second species), reduction and deformation of ornamentation (Khanna et al., 2013), and increased pseudopore area (Knorr et al., 2015). That foraminifera can form under acidified conditions is no assurance of their persistence, with some studies indicating that dissolution can increase under acidification (Khanna et al., 2013; McIntyre-Wressnig et al., 2013), potentially reducing the number of tests (Haynert and Schönfeld, 2014).

Effects of ocean acidification on particular foraminifera have the potential to combine to modify overall communities. Such potential has been illustrated in a study of a volcanic vent system in Italy where pCO_2 is enriched near the vents, and which has revealed marked foraminiferal distribution and diversity change along a gradient of pH (Dias et al., 2010). Arctic communities may shift in similar ways under ocean acidification (see further detail on species interactions in Section 3.3).

3.2.5 Macroalgae

3.2.5.1 Calcifying macroalgae

Ocean acidification is anticipated to have adverse effects on algal calcification in a range of macroalgae. Coralline algae are often proposed to be particularly susceptible to ocean acidification given that their skeletons include a form of calcium carbonate (high magnesium calcite) that readily undergoes dissolution under low pH (McCoy and Kamenos, 2015; Cornwall et al., 2017a). Recent work looking at coralline algal species, including those found within Arctic ecosystems, has corroborated previous foundational work considering the effects of elevated pCO_2 on coralline algal physiology and growth. These previous studies indicate a parabolic growth response to pH and pCO_2 (reviewed by McCoy and Kamenos, 2015). Within colder areas, where a Svalbard population of *Lithothamnion glaciale* was considered, the relative net calcification rate decreased under elevated pCO_2 suggesting that conditions are currently near the peak of the parabola, with any change sufficient to drive a shift past a tipping point such that a reduction is observed (Büdenbender et al., 2011). The response observed can, however, be dependent upon the experimental period considered; in laboratory cultures *L. glaciale* maintained growth rates when exposed for three months (Ragazzola et al., 2012), but rates were reduced if exposed for 10 months (Ragazzola et al., 2013). This reduction was suggested to result from a reallocation of the energy budget over this period, highlighting a high degree of plasticity (Ragazzola et al., 2013).

Ocean acidification can modify the structure of the calcified parts produced. As with rate of calcification, this can depend on the experimental period considered; in laboratory cultures of *L. glaciale* skeletal quality (intra- and inter-cellular wall thickness) was decreased when exposed for three months (Ragazzola et al., 2012), but preserved if exposed for 10 months (Ragazzola et al., 2013). Moreover, under ocean acidification the geochemistry of structures can be modified. *L. glaciale* grown under elevated pCO_2 conditions lacked magnesium banding (while this occurred in the controls), and overall magnesium concentrations were lower than in the control, potentially altering structural properties of the algae by reducing elasticity (Ragazzola et al., 2016).

The responses of other processes to ocean acidification have also been considered, such as the ability to produce the secondary metabolite dimethylsulfoniopropionate (DMSP) that acts as a cryoprotectant, antioxidant, and possible grazer defense compound (references in McCoy and Kamenos, 2015). Where an acidification scenario has been considered (exposure to

~1080 μatm pCO_2), the Arctic algae *L. glaciale* has shown no change in the production of DMSP (Burdett et al., 2012).

Calcification of brown and green algae has been considered in response to ocean acidification, albeit not in the Arctic. Although brown algae are not obligate calcifiers, they do produce calcium carbonate. A study of temperate and tropical brown algae identified that they showed reductions in calcium carbonate content with CO_2 enrichment, yet the algae did increase in abundance (Johnson et al., 2012). Studies on temperate green algae have identified that at high CO_2 they are less calcified, less stiff, and droopier; indicating changes in skeletal performance (Newcomb et al., 2015).

3.2.5.2 Non-calcifying macroalgae

Several non-calcifying macroalgae have been shown to benefit from ocean acidification (Hall-Spencer et al., 2008; Koch et al., 2013; Cornwall et al., 2017b). Experimental evidence from geographically diverse studies shows species- and location-specificity in responses (e.g., Hepburn et al., 2011; Falkenberg et al., 2013; Celis-Plá et al., 2015). This variability is likely to be reflected in the responses of species found in the Arctic.

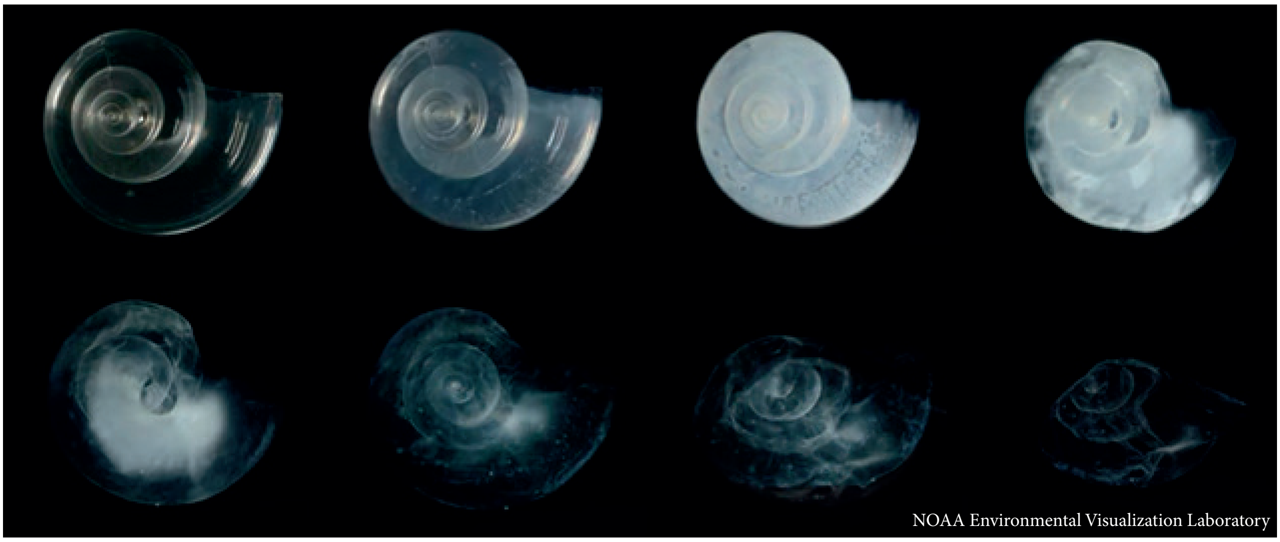
Ocean acidification may modify photosynthesis in macroalgae. Given that most macroalgae use carbon concentrating mechanisms (CCMs) to increase CO_2 concentration at the site of photosynthesis, they are unlikely to be carbon-limited under current conditions (Raven et al., 2008; Koch et al., 2013; Cornwall et al., 2017b). The use of CCMs is, however, energetically costly (Raven et al., 2014). An Arctic population of the kelp *Saccharina latissima* exposed to high pCO_2 deactivated CCMs while the Rubisco content remained unaltered, a change associated with an increased growth rate (Olischläger et al., 2017). In this study, algae from a cold-temperate area were also considered; this ecotype showed different responses compared to the Arctic ecotype, indicating that the Arctic population is more likely to benefit under ocean acidification and highlighting that responses may be location-specific and not easily transferrable. Elsewhere in the Arctic, high pCO_2 has been found to increase the growth rates of *S. latissima* and *Alaria esculenta* (Gordillo et al., 2015; Iñiguez et al., 2016). It has, however, been found that both *S. latissima* and *Laminaria solidungula* may be largely unaffected by increased pCO_2; photosynthesis was indicated to be carbon saturated at current levels as enriched pCO_2 did not influence carbon fixation, and no deactivation of CCMs was suggested by the ^{13}C isotopic discrimination values. Moreover, there was no change in algal growth rate associated with the manipulated pCO_2 conditions (Iñiguez et al., 2016). This species-specificity is highlighted in a single study where six common species of Arctic macroalgae were considered; one species responded positively (*Saccorhiza dermatodea*), no change was observed in four species (*Monostroma arcticum*, *Phycodrys rubens*, *Ptilota plumosa*, *Alaria esulenta*), and one species responded negatively (*Desmarestia aculeata*) (Gordillo et al., 2016) (see also Iñiguez et al., 2016).

The biochemical composition of macroalgae may be modified under ocean acidification. In considering six species of macroalgae found in the Arctic, Gordillo et al. (2016) identified that pCO_2 mainly modified the internal accumulation of carbohydrates and lipids, while the C:N balance was largely unaffected. In terms of carbohydrates, elevated pCO_2 led to an increase in *Monostroma arcticum*, *Phycodrys rubens*, *Desmarestia aculeata*, and *Saccorhiza dermatodea*, a decrease in *Ptilota plumosa*, and no change in *Alaria esculenta*. Elevated pCO_2 increased lipid content in *M. arcticum* and *D. aculeata*, decreased it in *P. rubens*, and led to no change in *S. dermatodea*, *P. plumose*, or *A. esculenta*.

The species-specific responses are of significance given that those species which best tolerate or benefit from ocean acidification are likely to become dominant, potentially leading to major shifts in macroalgal composition in marine systems, including those of the Arctic (Porzio et al., 2011; Connell et al., 2013). Moreover, changes in algal traits can influence their interactions with the organisms that consume them, such as sea urchins (discussed further in Annex 2).

3.2.6 Corals

Coral reefs can form complex structures in the Arctic, with the largest known deep- and cold-water coral reef composed of *Lophelia pertusa* and *Madrepora oculata* and located north of the Arctic Circle off Norway at a depth of 300–400 m (Sabatier et al., 2012). Under short-term ocean acidification, *L. pertusa* from southern Norway showed only slight reductions in net calcification rates, while significantly elevating respiration and capture rates of prey (specifically *Artemia salina*) (Georgian et al., 2016). Although there can be a negative shock response in the short term, under longer-term exposures of weeks to months *L. pertusa* from a range of locations have been found to maintain calcification rates (e.g., Form and Riebesell 2012; Maier et al., 2013a,b; Hennige et al., 2014, 2015; Movilla et al., 2014), suggesting that Arctic *L. pertusa* may also be able to calcify under persistent ocean acidification if they have enough food to meet their metabolic costs (Rodolfo-Metalpa et al., 2015). The cold-water coral *M. oculata* has also been found to be able to maintain calcification rates under ocean acidification scenarios in a range of locations (e.g., Movilla et al., 2014). It is worth noting, however, that negative responses have been found in some conditions (Maier et al., 2016). The generally continued ability of these cold-water corals to calcify under different conditions is likely to be due to the strong upregulation of pH and consequent elevation of the internal carbonate saturation state. This modulation of the proton gradient between seawater and the site of calcification is attributed to the action of Ca^{2+}-ATPase (McCulloch et al., 2012). The acclimation of coral species to longer-term increased pCO_2 (i.e., over 12 months) such that net calcification is maintained is, however, associated with other physiological consequences. For example, the process of internal pH upregulation has an energetic cost, and can therefore affect growth rates (McCulloch et al., 2012). In addition, under elevated pCO_2 the skeletal structure may be changed such that it exhibits decreased crystallographic and molecular-scale bonding organization, affecting breaking strength (Hennige et al., 2015). Some corals have, however, shown no distinctive differences between natural and low pH conditions in terms of skeletal morphology, macro-morphological skeletal arrangement, or secondary thickening (Wall et al., 2015). Understanding


NOAA Environmental Visualization Laboratory

In laboratory experiments, this pterapod shell dissolved over the course of 45 days in seawater adjusted to an ocean chemistry projected for the year 2100.

whether structural changes will occur under acidification is important as it influences the weakness of the corals, potentially making them more susceptible to bioerosion and mechanical damage (Hennige et al., 2015).

While the text has focused here on living corals, it is worth noting that much of these habitats comprises dead coral skeletons that are vulnerable to dissolution as a consequence of ocean acidification. For example, where the coral colonies of reefs located off the coast of Scotland were examined in terms of dead/living tissue, it was revealed that at least 73% of the *Lophelia pertusa* colonies were composed of exposed dead coral skeleton (Vad et al., 2017). Concern exists, therefore, that ocean acidification may cause dissolution and collapse of deep water reefs as shoaling of the aragonite saturation horizon exposes them to corrosive waters (Jackson et al., 2014). It is possible that if the dead zone at the bottom of the coral disintegrates under ocean acidification, then the entire structure would be in danger of collapse.

3.2.7 Mollusks

3.2.7.1 Gastropods

Of the gastropods, pteropods have been suggested to be particularly sensitive to forecasted ocean acidification as their shells are made of aragonite, a relatively soluble form of biogenic calcium carbonate (Manno et al., 2017). Many of the studies considering the response of the Arctic members of this group have focused on *Limacina helicina*. Under ocean acidification, these pteropods have demonstrated reduced shell size (Lischka et al., 2011), reduced shell extension (Comeau et al., 2012), decreased precipitation of calcium carbonate (correlated to the aragonite saturation state) (Comeau et al., 2010), and increased shell degradation (Lischka et al., 2011; Lischka and Riebesell, 2012). The species *L. retroversa* has shown similar patterns of increased shell degradation under ocean acidification (Lischka and Riebesell, 2012). Unrelated to calcification or shells, under acidification *L. helicina* has also shown increased mortality (Lischka et al., 2011, but no effect according to Comeau et al., 2012), and lower egg organogenesis (Manno et al., 2016), but unchanged respiration rates (Comeau et al., 2010), and unchanged gut clearance rates (Comeau et al., 2010). A molecular approach has been applied to *L. helicina* and identified that genes were both upregulated and downregulated in response to ocean acidification (Koh et al., 2015). Consideration of another polar pteropod, *Clinoe limacina*, revealed that of 300,994 transcripts, 41 were differentially expressed following an ocean acidification treatment with 28 upregulated and 13 downregulated. The authors suggested that this may correspond with limited physiological responses of the species to short-term exposure (three days) (Thabet et al., 2017).

An array of marine gastropods found in the Arctic have been considered in the context of ocean acidification – including limpets, abalone, and littorinids. While limpets are unstudied in the Arctic, they have been considered in other regions. These studies include measurements along natural gradients of increasing $p\text{CO}_2$, where they have been found to be severely impacted by the modified environment (Rodolfo-Metalpa et al., 2011; Garilli et al., 2015). For some limpets, reduced pH has had no effect on mortality (Maboloc and Chan 2017). However, as for other groups, traits associated with calcification have been significantly affected, including negative effects on the process of calcification (Noisette et al., 2016) (but see Schram et al., 2016 for an example of no effect), reduced growth rates (Maboloc and Chan, 2017), smaller shell sizes (Maboloc and Chan, 2017), and increased porosity (Maboloc and Chan, 2017). Experimental results have suggested that some limpets that experience increased dissolution under acidification may be able to counter this through shell repair which must have a metabolic cost (Langer et al., 2014). In contrast, other physiological patterns and processes appear unaffected by acidification, such as respiration, ammonia excretion, and filtration (Noisette et al., 2016) and lipid allocation in reproductive organs (Schram et al., 2016). The behavior of these organisms under ocean acidification has also been considered, with their capacity to right (when overturned) and mean maximal escape speed found to be unchanged (Schram et al., 2014).

Abalone responses to ocean acidification have often been studied in their larval and juvenile stages. Processes in young abalone seem to be particularly susceptible to acidification, with negative effects on fertilization rate (Guo et al., 2015), hatching rate (Guo et al., 2015), trochophore development (Guo et al., 2015), veliger survival (Guo et al., 2015), metamorphosis (Guo et al., 2015, but see Crim et al., 2011), larval survival (Crim et al., 2011), shell development (i.e., abnormalities/lacking shell) (Byrne et al., 2011; Crim et al., 2011), shell size (Crim et al., 2011; Cunningham et al., 2016), wet weight (Cunningham et al., 2016), and shell weight (Cunningham et al., 2016) (but expression of shell formation genes was found to be unchanged by Zippay and Hofmann 2010). It is worth noting that effects of ocean acidification have also been identified in other gastropod groups not detailed here such as littorinids (Ellis et al., 2009). That many processes in larval and juvenile gastropods, including abalone, have been negatively affected provides support for the common suggestion that these stages may be particularly susceptible to ocean acidification (e.g., Byrne, 2011).

3.2.7.2 Bivalves

A key group of mollusks are bivalves, which include clams, oysters, and scallops. The responses of the clams *Macoma calcarea*, *Astarte montagui*, and *A. borealis* from the Pacific Arctic have been experimentally investigated in the context of ocean acidification. At the completion of the experiment, it was found that the shells of *A. borealis* showed a decrease in length, while those of the other species were unaffected. In addition, wet weight and oxygen consumption were not significantly different for any of the species, although there was a trend for these features to be negatively affected (Goethel et al., 2017). Other Arctic bivalves, *Chlamys islandica* and *Ciliatocardium ciliatum*, have also been investigated in a field study where there was a subtle difference in the aragonitic content of shells linked to depth and, consequently, also with water ion concentration, pH and $p\text{CO}_2$ (Iglikowska et al., 2017). An Antarctic bivalve, *Laternula elliptica*, has been found to have slowed development of calcifying stages (Bylenga et al., 2015), and modified ultrastructure of the larvae (e.g., shape, edges, hinges, surfaces) under acidification (Bylenga et al., 2017).

Extensive research considering mollusk responses to ocean acidification has been conducted in non-Arctic areas, facilitating the production of review papers for some groups. In terms of oysters, for example, a review of existing literature revealed that under ocean acidification adult oysters typically have reduced growth and calcification rates, while larval oysters display stunted growth, developmental abnormalities, and increased mortality (Lemasson et al., 2017). A review of scallop literature highlighted that although this group may be adversely affected under ocean acidification, the tolerance of particular species will be determined by their structure, life history, environmental preferences, behavior, physiology, and sources of nutrition (Richards et al., 2015).

3.2.7.3 Cephalopods

The response of cephalopods to ocean acidification has been considered in a range of geographical areas, with potential that similar responses may be observed in the Arctic. As with other mollusks, there has been focus placed on identifying effects of ocean acidification on cephalopod features and processes such as metabolic rates (unchanged, Rosa et al., 2013; reduced, Rosa and Seibel, 2008; Hu et al., 2014b), hatching (increased hatching time, Kaplan et al., 2013; Sigwart et al., 2016; increased premature hatching, Rosa et al., 2013), growth (unaffected, Dorey et al., 2013; reduced, Sigwart et al., 2016), size (reduced, Kaplan et al., 2013; Sigwart et al., 2016), and survival (unchanged, Rosa et al., 2013). In contrast to many other species with calcified structures, the internal shell of the cuttlefish (or the cuttlebone) has displayed features indicative of hypercalcification under ocean acidification (Dorey et al., 2013), enabling juveniles to maintain this process (Gutowska et al., 2008). This calcified structure can, however, have modified morphology under acidification (i.e., altered spacing of laminae, pillar thickness) (Gutowska et al., 2010). Cephalopod activity and behavior has been found to be modified under acidification, with altered defensive behaviors (toward jet escapes and use of ink, reduction in use of defensive arm postures). Such changes would affect the energy budget of individual organisms, and modify interactions with predators (Spady et al., 2014).

3.2.8 Echinoderms

The echinoderms investigated in the context of ocean acidification include sea urchins, brittlestars, sea stars, and sea cucumbers. In the Arctic, the response of the green sea urchin, *Strongylocentrotus droebachiensis* has been studied (discussed further in Annex 2). Exposure of reproductive stages to acidified conditions identified that acidification can increase the proportion of eggs that fail fertilization, increase the risk of polyspermy (due to failures in fertilization envelope formation), and increase irregular formation of the embryo (due to impaired formation of the hyaline layer) (Bögner et al., 2014). Other studies considering urchins at polar locations, specifically in the Antarctic, have found that fertilization and early cell division may be largely resilient to acidification (Ericson et al., 2010, 2012; Yu et al., 2013; Kapsenberg and Hofmann 2014), although there is potential for slightly greater sensitivity of later development through to gastrula (Ericson et al., 2010), reduction in the percentage of normal embryos (Ericson et al., 2012), or a slight delay in hatching (Yu et al., 2013). The larvae produced under acidification may grow more slowly (Byrne et al., 2013), develop shorter arms (a highly plastic morphological aspect) (Byrne et al., 2013; Yu et al., 2013), have disrupted developmental patterning as indicated by increased left-right asymmetry and altered body allometry (Byrne et al., 2013), or be smaller in terms of body component (Clark et al., 2009).

The Arctic brittlestar, *Ophiocten sericeum*, has been investigated in the context of ocean acidification, with metabolism upregulated, an unchanged number of muscle nuclei and no change in arm regeneration (Wood et al., 2011). Other brittlestars have shown little change in terms of oxygen consumption (Wood et al., 2010), metabolism (Wood et al., 2008), mobility (Wood et al., 2010), and the percentage of calcium and magnesium in arm (Wood et al., 2010). It is worth noting, however, that ocean acidification has been linked to increased larval mortality (Dupont et al., 2008; Chan et al., 2015), reduced larval swimming speeds (Chan et al., 2015), abnormal development (Dupont et al., 2008), skeletogenesis (Dupont et al., 2008), reduced arm regeneration (Hu et al., 2014a), considerable muscle wastage (Wood et al., 2008), reduced metabolic rates (potentially reflecting uncompensated acidosis) (Hu et al., 2014a), and increased ammonium excretion rates (Hu et al., 2014a). A gene-expression analysis of brittlestars revealed that there may also be reduced expression of acid–base and metabolic genes (Hu et al., 2014a).

Sea stars from polar environments have largely been found to respond negatively to ocean acidification. That is, while reduced pH may have little effect on fertilization (Gonzalez-Bernat et al., 2013), it can affect larval survival (i.e., reduced), development, and morphology (i.e., shape and size) (Gonzalez-Bernat et al., 2013). When exposed to acidified conditions, adult sea stars from polar regions experienced extracellular acidosis, which remained uncompensated within a period of seven days. The coelomic fluid acidosis was associated with an increase in total coelomocyte number (Dupont and Thorndyke, 2012).

When exposed to forecasted acidification, the cold-water sea cucumber *Cucumaria frondosa* has shown impaired gamete synthesis, which has led to discrepancies in oocyte/embryo buoyancy, morphology, and developmental tempo, translating to increased mortality before the blastula stage. There were also differences in the microstructural appearance of ossicles and lipid contents of muscles, gonads, and spawned oocytes (Verkaik et al., 2016). Under acidification other sea cucumbers have demonstrated reduced specific growth rates, reduced energy consumption and defecation rates, and shifted energy budgets (resulting in a lowered allocation to somatic growth) (Yuan et al., 2016). In terms of reproduction, sperm flagellar motility was significantly reduced under acidification scenarios (Morita et al., 2010). Reduced pH did, however, have relatively small effects on the sea cucumber relative to other echinoderms, despite this change leading to a decrease in post-fertilization success and subtle differences in growth and development (specifically stage duration) (Yuan et al., 2015).

3.2.9 Crustaceans

Of the Arctic crustaceans, copepods are perhaps the most ecologically important and well-studied in the context of ocean acidification. A particular focus has been placed on *Calanus* spp., specifically the Arctic copepod *Calanus glacialis*. In this species, ocean acidification effects vary with life stage. The developmental rate of nauplius larvae appears largely unaffected by acidification, probably as a result of physiological buffering by changes to the universal stress response (including DNA repair, redox regulation, protein folding, proteolysis) and upregulation of cellular ion transport, particularly sodium/proton antiporters (Bailey et al., 2016, 2017). In contrast, the copepodite stages seem more sensitive. In the early copepodite stages (CII-CIII), ocean acidification seems to induce increased costs of biosynthesis (Thor et al., 2016). In copepodites from Kongsfjord, West Svalbard, Thor et al. (2016) found a 2.5 times greater increase in metabolic rates due to feeding at elevated pCO_2. Further studies have shown that the relationship between metabolic rate and ingestion rate is similarly affected in the later CIV stage; in *C. glacialis* from two fjords on the Svalbard west coast scope for growth (a measure of the energy available for growth calculated as ingestion rate times gut absorption efficiency minus metabolic rate) decreased by up to 50% under increased pCO_2 (Thor et al., 2018a). Such changes to both early and late copepodite stages would have serious implications for *C. glacialis* populations. Specifically, reductions in scope for growth would prolong stage development time and reduce the individual body size of developing copepodites and ultimately also reduce adult body size. In contrast to the effects on earlier copepodite stages, the last copepodite stage (CV) seems unresponsive to increased pCO_2. Several studies have shown no effects on rates of ingestion and metabolism in this stage (Hildebrandt et al., 2014, 2016; Thor et al., 2016). This non-response probably occurs as CV copepodites are metabolically different compared to the earlier stages. That is, while somatic growth is the main response in the preceding stages, metabolism is largely reconfigured to accommodate overwintering diapause in CVs. During diapause, *C. glacialis* CVs experience extracellular pH as low as 5.5 (possibly due to metabolic depression during hibernation) (Freese et al., 2015). In adult *C. glacialis*, fecundity also seems unaffected by high pCO_2 both in terms of egg production and egg hatching success and timing (Weydmann et al., 2012; Thor et al., 2018b).

Assessing the effects of ocean acidification in the context of the populations and communities that occur will be important in predicting the effects of ocean acidification. In terms of copepods, despite clear effects of ocean acidification on *Calanus glacialis* in laboratory studies, in more complex mesocosm-based communities deployed in Kongsfjord, West Svalbard, no differences have been found in stage development during the summer growth season (Niehoff et al., 2013). This mesocosm study also showed that copepod species composition did not change under acidification treatments. Thus, other effects may have countered the direct effects of increased pCO_2. For instance, acidification effects may have been mitigated by elevated food intake as primary production, and hence the availability of phytoplankton prey, increased with pCO_2 (Engel et al., 2013) (the potential role of these trophic links is further discussed in Section 3.3).

Arctic crabs appear sensitive to ocean acidification. In the spider crab (*Hyas araneus*) greatly increased pCO_2 (3000 ppm; in contrast to the current 380 ppm) caused increased development time and reduced survival of zoea I larvae in Kongsfjord (Walther et al., 2011; see also Schiffer et al., 2014). Similarly, even at a more moderate pCO_2 (710 ppm) effects were observed, although they were less pronounced (Walther et al., 2011). Larval physiological processes may be impaired in *H. araneus* exposed to acidification. Larvae from Kongsfjord showed lower capacity for calcium incorporation at high pCO_2 than those from other regions, suggesting that crab larvae developing at the cold end of the species distribution range may be more sensitive to ocean acidification than those in temperate regions (Walther et al., 2011). Adults have shown uncompensated extracellular acidosis at elevated pCO_2 potentially reducing muscular function, but there was no effect of ocean acidification on movement (Zittier et al., 2013). In the red king crab (*Paralithodes camtschaticus*) the survival of larvae has been shown to be compromised by elevated pCO_2, with 100% mortality occurring after 95 days in ~1600 µatm CO_2 water (Long et al., 2013). These detrimental effects also seem to extend to juveniles, with juveniles of both red king crab and tanner crab (*Chionoecetes bairdi*) from the Bering Sea experiencing decreased growth and condition index (also known as body mass index, which is dry mass in grams divided by the carapace length^3 for red king crab or carapace width^3 for tanner crab, in millimeters) and increased mortality under elevated pCO_2 (Long et al., 2013).

Zoea larva of *Homarus gammarus*.

The crab species *Carcinus maenas* is ecologically important in a range of regions, including the boreo-Arctic. Studies across the geographical range of this crab have indicated that acidification can affect it as detailed below. Acidification has, for example, been found to reduce feeding rates, and to prompt active extracellular pH compensation via bicarbonate accumulation (Appelhans et al., 2012). Quantification of acid–base regulation in the gills of this crab has shown that hemolymph K^+ concentrations, ammonia concentrations, and ammonia excretion rate were increased under elevated $p\mathrm{CO}_2$. Quantitative gene expression analysis revealed that under elevated $p\mathrm{CO}_2$ mRNA levels of transcripts hypothesized to be involved in ammonia and acid–base regulation showed varied responses, being upregulated in some individuals and downregulated in others (Fehsenfeld and Weihrauch, 2013). Another study identified that most of the genes known to code for proteins involved in osmo– and acid–base regulation and the cellular stress response were not impacted by elevated $p\mathrm{CO}_2$. In contrast, changes were observed in a calcium-activated chloride channel, a potassium channel, a tetraspanin, an integrin, a putative syntaxin-binding protein, and a $\mathrm{Cl}^-/\mathrm{HCO}_3^-$ exchanger (Fehsenfeld et al., 2011). Another study considering the gills, along with hemolymph and leg muscle, found that exposure to increased $p\mathrm{CO}_2$ led to changes in the metabolic profile, mainly due to a reduced level of intracellular osmolytes such as amino acids, potentially reflecting increased catabolism of amino acids to supply body fluids with proton-buffering ammonia (Hammer et al., 2012). Together these results suggest that some processes may be robust to ocean acidification, while others are likely to be more sensitive.

Decapods of importance in the Arctic include lobsters and shrimp. Studies of the lobster species *Homarus gammarus* from a range of regions often focus on early life history stages. In larvae, acidification was found not to affect survival, carapace length or zoeal progression, but did disrupt exoskeleton mineral content of the carapace (i.e., calcium and magnesium) and carapace mass (Arnold et al., 2009). In juveniles, acidification has been found to lead to increased mortality, typically due to molt death syndrome, as well as to reduced metabolism, food acquisition, and carapace mineral content (Small et al., 2016). A longer-term study considering both larval and juvenile exposure identified that while there was no clear effect of $p\mathrm{CO}_2$ on carapace length or dry weight, it did increase the deformities observed (e.g., curled carapace, damage in tail fan, bent rostrum, deformed claws, stiff/twisted walking legs, and puffy carapace). These morphological changes may then, in turn, influence a range of activities including respiration, ability to find food or sexual partners, and motility (Agnalt et al., 2013).

In northern shrimp (*Pandalus borealis*) larval development has typically been found to be negatively affected by increased $p\mathrm{CO}_2$. Studies on *P. borealis* from the Norwegian coast showed that while egg hatching is unaffected, all tested zoea larval stages (II, III, IV) developed significantly slower at ~1200 μatm CO_2 (Bechmann et al., 2011; Arnberg et al., 2013). These effects may be alleviated at higher temperature (increase from 6.7 to 9.5°C), so direct $p\mathrm{CO}_2$ effects may be masked by future climate change (Arnberg et al., 2013) (see Section 3.5 for further discussion of interactive effects of environmental changes). Adult shrimp may also be affected by elevated $p\mathrm{CO}_2$; one study found a 63% increase in adult mortality in *P. borealis* from the Swedish west coast (Dupont et al., 2014). Should this change in mortality also apply to Arctic *P. borealis* populations, then the consequences will be detrimental to the Arctic benthic community. (The effects of ocean acidification on *P. borealis* are explored further in a case study on the Greenland shrimp fishery; Annex 4.)

Arctic barnacles, although less studied than some other crustaceans, may be affected by acidification. For example, *Semibalanus balanoides* barnacles from the northern edge of the species' range in Svalbard were found to respond to lower pH; the growth and development of metamorphosing post-larvae were negatively impacted. It is important to note, however, that mineral composition was unaltered. This combination of responses indicates that there may have been a change in the energetic balance of the organisms with energy allocated to maintaining mineral integrity rather than growth (Findlay et al., 2010).

3.2.10 Other invertebrates

While the majority of research considering effects of ocean acidification on invertebrates has focused on the groups outlined in previous sections, representatives of other groups have also been studied. This section highlights some of these of relevance to the Arctic, selected using the qualitative review approach, specifically cnidarian, bryozoan, brachiopod, polychaete, and nematode. While few of the studies detailed below are from the Arctic, the responses observed elsewhere may inform predictions for this region.

A cnidarian found within the Arctic is the lion's mane jellyfish (*Cyanea capillata*). The effect of ocean acidification on scyphozoan polyps was investigated in a study where biological material was collected from Helgoland. Both polyp growth and carbon content were unaffected by the $p\mathrm{CO}_2$ treatments, indicating that this environmental change is unlikely to have direct effects on the growth of scyphisotomae (Lesniowski et al., 2015). Resilience of jellyfish to ocean acidification may have harmful outcomes such as increased blooms that can, in turn, have negative interactions with aquaculture (Hall-Spencer and Allen, 2015).

Reviews of bryozoan responses to ocean acidification have indicated that this group is likely to reduce calcification, change mineralogy, reduce growth, decrease survival, and lower

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A polar cod (*Boreogadus saida*) rests in an ice-covered space. Alaska, Beaufort Sea, North of Point Barrow.

production of polymorphic zooids for defense and reproduction (Smith, 2014; Taylor et al., 2015). However, Borszcz et al. (2013) found no evidence that increasing water depth (with assumed pH decrease) affected exoskeleton magnesium content for Arctic bryozoans, as might be expected on the basis that high-magnesium calcite is especially prone to dissolution.

While the response of Arctic brachiopods to ocean acidification has not been investigated, that of Antarctic brachiopods has been. The Antarctic brachiopod *Liothyrella uva* was found to have consistent rates of shell repair (with over 80% of all damaged individuals at the start of the experiment completing shell repair after 12 weeks) and growth under acidified scenarios (Cross et al., 2015). Another study of this brachiopod found that it suffered post-mortem shell dissolution under acidification (McClintock et al., 2009). In contrast, where live individuals of a New Zealand brachiopod *Calloria inconspicua* were investigated, they also showed a persistent capacity to repair shells under acidification, and were able to maintain (or even increase) their growth rate (Cross et al., 2016).

Within the annelids polychaete worms, including the ~200 Arctic species, have the potential to be impacted by ocean acidification. Experimental manipulation of pCO_2 has indicated that in the short term these worms can survive, increase their energy metabolism, and decrease carbonic anhydrase concentration; these homeostatic changes are, however, suggested to be unsustainable over the longer term (Turner et al., 2015). The activity of these organisms may be modified under acidification, as altered carbonate chemistry has been correlated with enhanced macroboring (Enochs et al., 2016). Reproduction may also be disrupted in polychaetes under ocean acidification with specific changes anticipated including larger and more abundant oocytes, fewer spermatozoa, lower effective fecundity (number of eggs laid), slower development of embryos and larvae (although the microstructure of the body wall, and chaeta appearance and elemental composition may be unaffected) (Verkaik et al., 2017). These effects may lead to the reduction of polychaete species richness and abundance under extreme low pH conditions, as has been observed in a volcanic vent system near Italy (Gambi et al., 2016).

Nematodes have been investigated in terms of their response to ocean acidification. Within meiofaunal assemblages, elevated pCO_2 can reduce nematode abundance and nematode species richness (Lee et al., 2017). Elevated pCO_2 has also been found to modify nematode abundance (decreased in Barry et al., 2004; Dashfield et al., 2008; increased in Hale et al., 2011; Meadows et al., 2015), with potential that nematode community structure and diversity will also be modified under acidified scenarios (Widdicombe et al., 2009; Meadows et al., 2015) (but see Dashfield et al., 2008).

3.2.11 Fishes

Ocean acidification has the potential to influence a range of physiological processes in fishes from Arctic waters, specifically polar cod (*Boreogadus saida*) and Atlantic cod (*Gadus morhua*) (Kunz et al., 2016). (Effects on *Gadus morhua* are discussed further in Annex 3). In general, adult marine teleosts are predicted to have low vulnerability to moderate acidification due to the capacities for acid–base regulation in their gills, intestine, and liver (Stapp et al., 2015; Hu et al., 2016; Kunz et al., 2016; Michael et al., 2016). Moreover, traits associated with mitochondrial acclimation potential (i.e., OXPHOS, proton leak, ATP production) have been found to be similar in groups acclimated to different pCO_2 conditions in both *B. saida* and *G. morhua* (Leo et al., 2017). It is worth noting, however, that these responses were modified under elevated temperature (Leo et al., 2017), an idea discussed further in Section 3.5.

Focus is increasingly being placed on reproductive stages and juvenile fish, given that they are potentially more susceptible than adults to the effects of ocean acidification. Experiments conducted on *Gadus morhua* have identified that while sperm swimming (which plays a central role in determining fertilization success) may be unchanged under acidification (Frommel et al., 2010), embryos experienced reduced hatching success, stimulated oxygen consumption, and reduced larval size at hatch, but no change in mitochondrial function or ionocyte abundance (Dahlke et al., 2017). Under ocean acidification, larval *G. morhua* exhibited higher mortality rates during the first 25 days following hatching, a critical phase for population recruitment (Stiasny et al., 2016). The otoliths (ear bones) of larval *G. morhua* have been found to show increased growth, with larger sagittae and lapilli (in terms of surface area normalized to fish length), although there was no significant difference in otolith shape or fluctuating asymmetry (difference between right and left sides) (Maneja et al., 2013b). Such changes in fish otoliths are important given that their role in neural and acoustic functions mean any change could represent an added mortality risk. Other tissues of the larvae, specifically those associated with internal organs, have been shown to be damaged under elevated pCO_2, with the degree of damage increasing with pCO_2 concentration (Frommel et al., 2012). Frommel et al. (2014) conducted a similar study on Atlantic herring (*Clupea harengus*) larvae exposed to pCO_2, finding that exposure to elevated pCO_2 resulted in stunted growth and development, decreased condition, and severe tissue damage in several organs. There have, however, also been reports of no effect of pCO_2 on a range of measures for *G. morhua* including hatching, survival, development, and otolith size (although it is worth noting this study was done using individuals from an environment already experiencing high ambient pCO_2) (Frommel et al., 2013).

Fish behavior may be modified under acidification. The behaviors displayed by tropical and temperate fish exposed to high pCO_2 include altered auditory performances, loss of lateralization (i.e., the preference for turning left or right), and changes in reproduction (Milazzo et al., 2016). In the context of the Arctic specifically, the behavior of *Boreogadus saida* and *Gadus morhua* have been investigated. While the activity of neither species was modified by ocean acidification, the behavioral laterality of *B. saida* was modified (i.e., the preference for one side over the other was reduced, and paralleled by a shift from right to left lateralization), while that of *G. morhua* remained unaltered (Schmidt et al., 2017a). The authors were prompted to conclude that fish in polar systems may undergo some, albeit less intense, behavioral disturbances under ocean acidification (Schmidt et al., 2017a). These changes may reflect that some brain metabolites in *B. saida* were modified by enhanced pCO_2 in isolation, while those of *G. morhua* remained unchanged (Schmidt et al., 2017b). The critical swimming speeds of *G. morhua* have also remained unchanged following acclimation to elevated pCO_2 (Melzner et al., 2009). The behavior of swimming larvae has been found to be consistent under ocean acidification; of the measured traits of swim duration, distance and speed, stop duration, and horizontal and vertical turn direction, it was identified that the only effects of high pCO_2 were restricted vertical turn angles, and reduced stop duration. These differences were subtle, with unclear functional and ecological significance (Maneja et al., 2013a). Similarly, in other regions, juvenile behaviors (including activity, emergence from shelter, relative lateralization, absolute lateralization, predator cue avoidance) appear robust under near future pCO_2 levels (Jutfelt and Hedgärde, 2013, 2015).

3.2.12 Seabirds and mammals

There is little evidence as to the direct effects of ocean acidification on seabirds and mammals in the Arctic. It has been suggested that any responses may be driven by indirect effects resulting from changes in quality of habitats or food resources (discussed further in Section 3.3), which can influence the movement, space use, energy budgets, and population abundance of seabirds and mammals (Jay et al., 2011; MacCracken 2012; Galbraith et al., 2014; Beatty et al., 2016; Thomas et al., 2016). In terms of change in habitat, the sensitivity of organisms to ocean acidification can be influenced by their distribution; the current patterns of occurrence may, for example, increase the susceptibility of certain whale populations to forecasted acidification (Thomas et al., 2016). As for the availability of food resources, species such as Pacific walrus (*Odobenus rosmarus divergens*) that feed on organisms potentially affected by ocean acidification (e.g., bivalves, gastropods, polychaetes), could be vulnerable to indirect effects driven by either food availability or food quality (Jay et al., 2011; Beatty et al., 2016).

3.3 Responses of ecosystems and habitats

In natural systems organisms interact with others in a range of ways. Ocean acidification has the potential to impact these interacting organisms differently, potentially shifting the ecosystem balance away from that which is currently observed (Hall-Spencer et al., 2008; Gaylord et al., 2015). Consequently, there is a growing body of research considering the responses of species assemblages and the habitats they form to ocean acidification.

Competitive interactions between different taxa may be affected by ocean acidification. For example, any change to calcifying algae could affect its competitive interactions with other space-occupying organisms (McCoy and Kamenos, 2015). That is, ocean acidification is likely to favor recruitment of fleshy, non-calcifying algae over calcified algae, potentially driving a shift from coralline algae (and the kelp they facilitate) to simple mat-algal dominated habitats (Porzio et al., 2011; Brodie et al., 2014). Similarly, ocean acidification may also cause loss of biogenic habitat complexity and a decline in species biodiversity and ecosystem function in other benthic habitats such as maerl beds, cold-water coral, and mussel reefs (Sunday et al., 2017).

Ocean acidification can have contrasting effects on organisms at different trophic levels, potentially modifying their interactions (i.e., that between prey and predator). For example, an experiment investigated the responses of pelagic communities including grazing copepods. It was identified that the interaction between copepods and dinoflagellate cell abundance was altered under acidification; copepods showed a stronger preference for dinoflagellates under elevated pCO_2 conditions, indicating changes in food quality and grazing selectivity (Tarling et al., 2016). Moreover, these interactions could be altered where the susceptibility of organisms to predation under ocean acidification modifies their ability to produce defensive structures, such as shells of gastropods (Bibby et al., 2007). Changes in trophic interactions can then influence the consuming organism – for example, in jellyfish food availability can modify the formation of polyps, if this food source is modified by pCO_2 the effect could then move further through the food chain (Lesniowski et al., 2015). Specifically, changes to lower-level organisms such as bivalves or mollusks driven by ocean acidification could affect larger, benthivorous predators such as Pacific walrus (*Odobenus rosmarus divergens*), bearded seal (*Erignathus barbatus*), and diving seaducks, such as spectacled eider (*Somateria fischeri*) (Goethel et al., 2017).

The host-parasite relationship can regulate individuals, populations, and communities, and is sensitive to changes in conditions of the surrounding environment. Relative to other interactions, such as the trophic interactions discussed above, parasitism has received little attention in the context of ocean acidification (MacLeod and Poulin, 2012). It has been suggested that as water chemistry changes, sympatric marine species will exhibit differential tolerances which could unbalance otherwise stable community dynamics (MacLeod, 2017). To date, research on the potential effects of ocean acidification has largely centered on gastropod hosts and trematode parasites in New Zealand (MacLeod, 2017), with future research needing to consider a wider range of species over larger geographical areas, including the Arctic.

Ocean acidification-driven changes to marine organisms and ecosystems can affect human societies. This is because humans rely on marine species and ecosystems for a range of services including provisioning (e.g., nutrition, materials, energy), regulation and maintenance (mediation of waste, toxics, and other nuisances; mediation of flows; mediation of physical, chemical, and biological conditions), and cultural (physical, experiential, spiritual and symbolic interactions). Ocean acidification has the potential to modify the availability of these resources which would, in turn, affect socio-economic systems (reviewed by Falkenberg and Tubb, 2017; examples provided in Annexes 5 and 6). While such reliance is widespread, these changes are likely to be felt particularly by Indigenous populations that are heavily dependent upon marine-derived resources. For example, the Pikialasorsuaq ecosystem is the most biologically productive region north of the Arctic Circle, and has supported Inuit for millennia; it is a region whose biota is critical for food, cultural, and spiritual resources (Eegeesiak et al., 2017). If ocean acidification does modify the productivity of these ecosystems, it could affect the health of the dependent regional Inuit community. Indigenous knowledge suggests that the recent, and possibly also future, environmental change and challenges to their resilience are greater than have been faced in the past (Turner and Clifton, 2009). Such susceptibility is likely to be widespread given that similar connections occur in other Arctic regions and communities.

In addition to modifying interactions with other species, changes in biota can feed back to modify the abiotic system within which the ecosystem occurs. When investigated in the context of phytoplankton, for example, increasing picoeukaryote abundance under ocean acidification could result in shallower organic matter remineralization in the future Arctic Ocean (De La Rocha and Passow, 2014). Similarly, declining coccolithophore abundances could provide less calcium carbonate for organic matter ballasting, which again should tend to shift remineralization depths towards the surface (De La Rocha and Passow, 2014). More generally, increased primary production and respiration in the oceans could increase pCO_2 and decrease their ability to act as a sink for atmospheric pCO_2 (Mostofa, 2016). These processes would constitute a feedback mechanism to increasing atmospheric CO_2 levels.

3.4 Acclimation and adaptation

The acclimation (short-term phenotypic plasticity, physiological changes in individuals) and adaptation (longer-term genotypic change, at population scale) potential of the different groups considered can influence their response to modified environmental conditions. In terms of acclimation, there have been studies considering organisms from corals (Form and Riebesell, 2012) to fish (Leo et al., 2017). Indeed, the majority of studies included here consider the response of individual organisms to changes in their environment over time-scales from days (e.g., Dupont and Thorndyke, 2012; Monier et al., 2014; Thabet et al., 2017) to months (e.g., Ragazzola et al., 2013). Less well-investigated, however, is the potential for adaptation. The adaptive potential of the different groups will be proportional to the population size and generation time, with groups that have large population sizes (conferring greater genetic variation) and short generation times having the highest adaptation rates (Riebesell and Gattuso, 2015). Indeed, it has been identified that adaptation under ocean acidification can be observed in plankton after a few hundred generations (Lohbeck et al., 2012; Schaum et al., 2013). Similarly, in copepods natural selection has been shown to alleviate severe ocean acidification effects after just two generations in *Pseudocalanus acuspes*. That is, while enhanced pCO_2 reduced egg production by nearly 70% in a naïve population, this reduction was diminished to 30% after exposure to high pCO_2 during two generations due to selection in genes related to RNA transcription control (Thor and Dupont, 2015; De Wit et al., 2016). It does, however, remain unclear whether the process can occur rapidly enough to keep ecosystem functions and services unchanged in the face of the forecasted rapid ocean acidification (Riebesell et al., 2013; Sunday et al., 2014). Trans-generational responses to modified conditions may also occur as a result of epigenetic effects, whereby phenotypic variations are inherited across generations without any variations in DNA (Chakravarti et al., 2016). For example, epigenetic effects have been proposed as the mechanism by which a clownfish (*Amphiprion melanopus*) exposed to ocean acidification improved offspring performance, probably through the activation of more efficient physiological pathways (Miller et al., 2012).

3.5 Interactive effects in a multi-stressor environment

Human activities are having a range of impacts on natural systems that extend beyond ocean acidification. Thus, it is necessary to consider their potential combined impacts. A condition being modified simultaneously with carbonate chemistry is temperature; future warming is anticipated in the Arctic (Stocker, 2014). A meta-analysis of global trends has identified that there is a trend toward enhanced sensitivity to ocean acidification where taxa are also exposed to warming (Kroeker et al., 2013). Where the responses of Arctic phytoplankton to acidification and warming have been investigated, the effect of increased $p\mathrm{CO_2}$ has been found to be modified by elevated temperature (Coello-Camba et al., 2014; Holding et al., 2015). Studies considering Arctic macroalgae typically highlight that there will be species-specific responses to ocean warming and acidification, with some species affected by one stressor and not the other, while others show a synergistic response (moreover, these responses can be process-specific) (Gordillo et al., 2016; Iñiguez et al., 2016; Olischläger et al., 2017). That a synergistic response was identified fits with expectations derived from other systems; a meta-analysis identified that synergistic interactions between environmental conditions are more common than additive or antagonistic interactions (Przeslawski et al., 2015).

The species-specific nature of responses to ocean acidification and warming could increase the probability of community change as a consequence of differential change to the performance of species (Gordillo et al., 2016; Iñiguez et al., 2016). Consideration of Arctic pteropod responses to acidification and warming revealed that they too are likely to be species-specific; for example in the study by Lischka and Riebesell (2012) while there was a significant $p\mathrm{CO_2}$ effect on shell degradation in two species investigated (the polar *Limacina helicina* and the boreal *L. retroversa*), synergistic effects between temperature and $p\mathrm{CO_2}$ were identified for *L. helicina*, indicating that polar species may, indeed, be particularly sensitive to warming and acidification. Where investigated for fish, acidification and warming tend to combine to have synergistic effects, resulting in more pronounced acidification effects at higher temperatures (e.g., Dahlke et al., 2017; Leo et al., 2017). Modelling approaches have indicated that under future climate conditions (i.e., acidification and warming) there may, indeed, be reduced future recruitment success for Atlantic cod (Koenigstein et al., 2018). Such interactions should be investigated in future experiments on species composition, especially in the Arctic where warming is projected to be relatively large (Stocker, 2014). These experiments and models need to consider species' mobility and current distributions as they can modify the importance of the interaction between warming and ocean acidification. Specifically, populations may be able to relocate latitudinally to maintain current temperature ranges, but then other environmental constraints may apply.

Alterations in underwater light regimes associated with climate change (Stocker, 2014) can modify the response of species to ocean acidification. Increased irradiance may result from sea-ice retreat associated with climate change. Experimental manipulations have shown that phytoplankton may be resistant to modified light and acidification when manipulated both in isolation and combination, challenging the common prediction of stimulatory effects on primary production (Hoppe et al., 2018). Macroalgae, however, have been found to respond in the widely anticipated manner, with two kelps synergistically benefiting from light enrichment and acidification. The highest growth rates were recorded under modification of both conditions in both species, although one did benefit more strongly than the other, highlighting that even where positive effects are observed species interactions can be modified by environmental change (Gordillo et al., 2015).

A broad range of environmental conditions can combine with acidification to modify organism response; these include copper exposure (Marques et al., 2017), phosphate limitation (Maat et al., 2014), and eutrophication (Reymond et al., 2013). Moreover, although experimental designs commonly consider two factors in a crossed design, more than two are being altered in Arctic environments. Consequently, experiments are also moving to consider more complex interactions, such as ocean acidification, temperature, and salinity (Haynert and Schönfeld, 2014), or ocean acidification, nutrients and light (Celis-Plá et al., 2015). Consideration of these complex interactions will require careful selection of an appropriate experimental strategy (Boyd et al., 2018).

Human activities are modifying the biotic environment, with these alterations set to combine with future global change in abiotic conditions. For example, humans are extracting key organisms, potentially restricting their range, while simultaneously facilitating the expansion of invasive species as new routes are opened for shipping and fishing, a change which is exacerbated as sea ice melts and allows ships through areas that were previously inaccessible (Brodie et al., 2014; Hall-Spencer and Allen, 2015). Where organism occurrence is modified, this can combine with ocean acidification to alter the occurrence of dependent higher-level trophic organisms (Lesniowski et al., 2015; Maier et al., 2016), or modify community composition (Dashfield et al., 2008). It will, therefore, be important to understand, and potentially manage, the biota that can modify (mediate or exacerbate) the effect of ocean acidification on other organisms (Falkenberg et al., 2012, 2014).

3.6 Conclusions

Ocean acidification has the potential to drive changes in Arctic marine systems. These changes can reflect direct impacts on the different groups considered, as well as the indirect effects that are mediated both within and between groups. There is likely to be great heterogeneity in the responses of organisms – with some positively influenced, others unaffected, and still more adversely impacted. Drawing generalized predictions of effects remains difficult given the species-, life stage-, location-, season-, etc. specificity of responses. Further complicating responses are changes humans are concomitantly driving in the abiotic environment (e.g., warming, altered light availability) and species assemblages (e.g., removal or introduction of key species). Where researchers begin to understand the mechanisms driving similarities and differences in responses, the capacity to draw more confident predictions and forecasts may develop. The results currently available do, however, indicate that forecasted ocean acidification is likely to be sufficient to drive changes in Arctic organisms and ecosystems to a magnitude that will affect the associated human societies.

Appendix: Manipulative experimental studies

Table A3.1 The manipulative experimental studies (i.e., excluding measures of natural systems, synthesis studies, modelling approaches etc.). Reported for each paper are the study species/community/assemblage, location from which the study species/community/assemblage was isolated or collected, and ocean acidification treatment (reported in either CO_2 as µatm or ppm, or pH) in terms of the ambient or control treatment (-) and modified treatments (reduced or increased ↓↑). Where possible, the treatment is reported as CO_2 (µatm or ppm), however, for some papers these conditions have been reported in terms of pH. If a single reference includes more than one study species/location/set of acidification treatments, this is indicated in the table (/).

Section / Source	Study species / community / assemblage		Location
3.2.1 / Viruses			
Crawfurd et al. 2017	Microbial community		Baltic Sea
Tsiola et al. 2017	Plankton community		Mediterranean Sea
Maat et al. 2014	Picoeukaryote	*Micromonas pusilla*	North Sea
Rochelle-Newall et al. 2004	Coccolithophore	*Emiliania huxleyi*	Norway
Celussi et al. 2017	Prokaryotic communities		Mediterranean Sea
Carreira et al. 2013	Coccolithophores (virus and host)	*Emiliania huxleyi*, *Phaeocystis poucheti*	Norway
Traving et al. 2013	Virus, cyanobacteria	*Cyanophage* S-PM2, *Syechonococcus* sp WH7803	English Channel, Sargasso Sea
Larsen et al. 2008	Virioplankton community		Norway
Chen et al. 2015	Virus, Algae	Viruses, *Phaeocystis globose*	China
3.2.2 / Bacteria and archaea			
Motegi et al. 2013	Bacterial community		Svalbard
Hornick et al. 2017	Bacteria-Phytoplankton community		Baltic Sea
Piontek et al. 2013	Bacterioplankton community		Svalbard
Piontek et al. 2015	Bacterioplankton community		Fram Strait
Sperling et al. 2013	Bacterioplankton community		
Zhang et al. 2013	Bacterioplankton community		Svalbard
Wang et al. 2016	Bacterioplankton community		Svalbard
Brussaard et al. 2013	Microbial community		Svalbard
Roy et al. 2013	Bacterial community		Svalbard
Tait et al. 2013	Microbial community		Arctic
Tait et al. 2014	Microbial community		Svalbard
Monier et al. 2014	Microbial community		Ellef Ringnes Island
Currie et al. 2017	Microbial community		UK
Hassenrück et al. 2016	Microbial community		Papua New Guinea
3.2.3 / Phytoplankton			
Pancic et al. 2015	Diatom	*Fragilariopsis cylindrus*	Greenland
Heiden et al. 2016	Diatom	*Fragilariopsis curta*, *Odontella weisflogii*	Antarctica
Wolf et al. 2018	Diatom	*Thalassiosira hyalina*	Norway

Ocean acidification treatment					
CO_2, µatm		CO_2, ppm		pH	
-	↓↑	-	↓↑	-	↓↑
365, 368	497, 821, 1007, 1231				
356±14	450, 550, 750, 840, 1000, 1250				
370	750				
		414	190, 714		
450 / 350	550, 650, 750, 850, 1000, 1250 / 450, 550, 750, 850, 1000, 1250				
		350	280, 700		
				8	7.6, 7
350	700, 1050				
390	1000				
185	270, 375, 480, 685, 820, 1050, 1420				
365, 368	497, 821, 1007, 1231				
175, 180	250, 340, 425, 600, 675, 860, 1085				
				~8.0	5 to 9
185	270, 375, 480, 685, 820, 1050, 1420				
175, 180	250, 340, 425, 600, 675, 860, 1085				
175, 180	250, 340, 450, 600, 675, 860, 1085				
185	270, 375, 480, 685, 820, 1050, 1420				
185	~ 270, 685, 820, 1050				
380	540, 750, 1120, 3000				
380	540, 760, 1120, 3000				
		400	880, 1800		
		380	750		
				8.24±0.02 / 8.33±0.00	7.83±0.08, 7.53±0.13 / 7.56±0.05, 6.78±0.02
				8	7.7, 7.4, 7.1
380	180, 1000				
370	180, 1000, 1400				

Table A3.1 cont.

Section / Source	Study species / community / assemblage		Location
Sett et al. 2014	Coccolithophore	*Emiliania huxleyi*, *Gephyrocapsa oceanica*	Norway, France
Kottmeier et al. 2016	Coccolithophore	*Emiliania huxleyi*	Pacific Ocean
Fu et al. 2007	Cyanobacteria	*Synechococcus*, *Prochlorococcus*	Sargasso and Mediterranean Seas
Maat et al. 2014	Picoeukaryote	*Micromonas pusilla*	North Sea
García-Gómez et al. 2016	Green algae	*Dunaliella tertiolecta*	Norway
Webb et al. 2016	Phytoplankton community		Baltic Sea
Hoppe et al. 2018	Phytoplankton community		Baffin Bay
Hoppe et al. 2017	Phytoplankton community		Davis Strait
Davidson et al. 2016	Microbial communities		Antarctica
Hussherr et al. 2017	Phytoplankton community		Arctic Ocean
Yoshimura et al. 2013	Plankton community		Bering Sea / Pacific
Thoisen et al. 2015	Phytoplankton community		West Greenland
Coello-Camba et al. 2014	Phytoplankton community		Arctic Ocean
Segovia et al. 2017	Plankton community		Norway
Schulz et al. 2017	Phytoplankton community		Norway
Bermudez et al. 2016a	Plankton community		Norway
Rossoll et al. 2012	Diatom, Copepod	*Thalassiosira pseudonana*, *Acartia tonsa*	Baltic Sea
Bermudez et al. 2016b	Plankton community		Baltic Sea
Wang et al. 2017	Plankton community		China
Taucher et al. 2017	Plankton community		Sweden
Garzke et al. 2016	Plankton community (copepods)		Baltic Sea
Cripps et al. 2016	Copepod	*Acartia tonsa*	UK
3.2.4 / Foraminifera			
Manno et al. 2012	Foraminifera	*Neogloboquadrina pachyderma*	Fram Strait
Davis et al. 2017	Foraminifera	*Globigerina bulloides*	USA
McIntyre-Wressnig et al. 2013	Foraminifera	*Amphistegina gibbosa*	USA
McIntyre-Wressnig et al. 2014	Foraminifera	*Bolivina argentea*, *Bulimina marginata*	USA
Marques et al. 2017	Foraminifera	*Amphistegina gibbosa*	Brazil
Sinutok et al. 2011	Algae, Foraminifera	*Halimeda macroloba*, *Halimeda cylindracea*, *Marginopora vertebralis*	Australia
Prazeres et al. 2015	Foraminifera	*Amphistegina lessonii*, *Marginopora vertebralis*	Australia
Fujita et al. 2011	Foraminifera	*Baculogypsina sphaerulata*, *Calcarina gaudichaudii*, *Amphisorus hemprichii*	Japan
Reymond et al. 2013	Foraminifera	*Marginopora rossi*	Australia
Vogel and Uthicke 2012	Foraminifera	*Amphistegina radiata*, *Heterostegina depressa*, *Marginopora vertebralis*	
Dissard et al. 2010	Foraminifera	*Ammonia tepida*	Wadden Sea

Ocean acidification treatment					
CO_2, µatm		CO_2, ppm		pH	
-	↓↑	-	↓↑	-	↓↑
N/A	~20-6000				
403±4	998±15				
		380	750		
370	750				
		390	900		
350	390, 840, 1120, 1400				
380	1000				
380	1000				
84	643, 1281, 1848, 1942, 2423				
N/A	~ 250-3300				
600 / 450	300, 960, 1190 / 230, 880, 1110				
				8	7.7, 7.4, 7.1
		380	1000		
390	900				
~ 300	395, 590, 890, 1165, 1425, 2060, 3045				
280	380, 560, 840, 1120, 1400, 2000, 3000				
380	740				
~347	up to ~1333				
400	1000				
N/A	~760				
560	1400				
400	1000				
		380	700		
				8.3	8.0, 7.7, 7.5
		410±30	1000, 2000		
		~420	1000, 2000		
				8.1	7.8, 7.5, 7.2
				8.1	7.9, 7.7, 7.4
430±23	855±30, 1168±36, 2015±20				
360	260, 580, 770, 970				
		380	700, 1000		
467±16 / 496±26	784±47, 1169±143, 1662±275 / 878±106, 1307±118, 1925±157				
		230	1900		

Table A3.1 cont.

Section / Source	Study species / community / assemblage		Location
Haynert and Schönfeld 2014	Foraminifera	*Ammonia aomoriensis*	Baltic Sea
Robbins et al. 2017	Foraminifera	*Amphistegina gibbosa*, *Archaias angulatus*	USA
Khanna et al. 2013	Foraminifera	*Haynesina germanica*	Scotland
Knorr et al. 2015	Foraminifera	*Archaias angulatus*	USA
3.2.5 / Macroalgae			
3.2.5.1 / Calcifying macrolgae			
Büdenbender et al. 2011	Coralline red algae	*Lithothamnion glaciale*	Svalbard
Ragazzola et al. 2012	Coralline red algae	*Lithothamnion glaciale*	Kattegat
Ragazzola et al. 2013	Coralline red algae	*Lithothamnion glaciale*	Kattegat
Ragazzola et al. 2016	Coralline red algae	*Lithothamnion glaciale*	Kattegat
Burdett et al. 2012	Coralline red algae	*Lithothamnion glaciale*	Scotland
3.2.5.2 / Non-calcifying macroalgae			
Falkenberg et al. 2013	Turf algae, Kelp	*Feldmannia* spp., *Ecklonia radiata*	Australia
Celis-Plá et al. 2015	Macroalgal assemblage		Italy
Olischläger et al. 2017	Kelp	*Saccharina latissima*	Spitsbergen, North Sea
Gordillo et al. 2015	Kelp	*Alaria esculenta*, *Saccharina latissima*	Svalbard
Iñiguez et al. 2016	Kelp	*Saccharina latissima*, *Laminaria solidungula*	Svalbard
Gordillo et al. 2016	Chlorophyte, Rhodophytes, Phaeophytes	*Monostroma arcticum*, *Phycodrys rubens*, *Ptilota plumosa*, *Alaria esculenta*, *Desmarestia aculeata*, *Saccorhiza dermatodea*	Svalbard
3.2.6 / Corals			
Georgian et al. 2016	Coral	*Lophelia pertusa*	USA / Mexico / Cuba, Norway
Form and Riebesell 2012	Coral	*Lophelia pertusa*	Norway
Maier et al. 2013a	Coral	*Madrepora oculata*, *Lophelia pertusa*	Mediterranean
Maier et al. 2013b	Coral	*Lophelia pertusa*, *Madrepora oculata*	Mediterranean
Hennige et al. 2014	Coral	*Lophelia pertusa*	Scotland
Movilla et al. 2014	Coral	*Lophelia pertusa*, *Madrepora oculata*	Mediterranean
Hennige et al. 2015	Coral	*Lophelia pertusa*	Scotland
Rodolfo-Metalpa et al. 2015	Coral	*Caryophyllia smithii*, *Dendrophyllia cornigera*, *Desmophyllum dianthus*	South Adriatic Sea, Malta, Ionian Sea
Maier et al. 2016	Coral	*Madrepora oculata*	Adriatic Sea
Wall et al. 2015	Coral	*Lophelia pertusa*	Norway
3.2.7 / Mollusks			
3.2.7.1 / Gastropods			
Lischka et al. 2011	Pteropod	*Limacina helicina*	Svalbard
Comeau et al. 2012	Pteropod	*Limacina helicina*	Canada Basin, Arctic Ocean
Comeau et al. 2010	Pteropod	*Limacina helicina*	Svalbard
Lischka and Riebesell 2012	Pteropod	*Limacina helicinaia*, *Limacina retroversa*	Svalbard
Manno et al. 2016	Pteropod	*Limacina helicina antarctica*	Southern Ocean

Ocean acidification treatment					
CO_2, µatm		CO_2, ppm		pH	
-	↓↑	-	↓↑	-	↓↑
566	1195, 2108, 3843				
				8	7.6
		380	750, 1000		
		480	1328		
		385±26 / 388±45	883±49, 989±57, 1573±89 / 754±80, 958±117, 1563±187		
422	589, 755, 1018				
408	566, 770, 1024				
422	589				
498±161	1081±488, 2778±4047				
		280-380	550-650		
500	700-800, 1200				
380	800, 1500				
		380	1000		
		390	1200		
		390	1100		
552±42 / 579±41	831±54, 1165±76 / 845±61, 1208±132				
~509	~605, 856, 981				
		400	280, 700, 1000		
		400	280, 700, 1000		
		380	750		
		384±23	809±61		
		380	750, 1000		
319	1058				
		400	800, 1600, 2000		
405	982±146				
		380	180, 750, 1150		
~380	~550, 760				
380	280, 550, 760, 1020				
350	650, 880				
		386	750, 1100		

Table A3.1 cont.

Section / Source	Study species / community / assemblage		Location
Koh et al. 2015	Pteropod	*Limacina helicina*	Svalbard
Thabet et al. 2017	Pteropod	*Clione limacina*	USA
Maboloc and Chan 2017	Limpet	*Crepidula onyx*	Hong Kong
Noisette et al. 2016	Limpet	*Crepidula fornicata*	France
Schram et al. 2016	Limpet, Snail	*Nacella concinna*, *Margarella antarctica*	Antarctica
Schram et al. 2014	Limpet, Mesograstropod snail	*Nacella concinna*, *Margarella antarctica*	Antarctic
Guo et al. 2015	Abalone, Oyster	*Haliotis diversicolor*, *Haliotis discus hannai*, *Crassostrea angulata*	China
Crim et al. 2011	Abalone	*Haliotis kamtschatkana*	Canada
Byrne et al. 2011	Abalone, Sea urchin	*Haliotis coccoradiata*, *Heliocidaris erythrogramma*	Australia
Cunningham et al. 2016	Abalone	*Haliotis iris*	New Zealand
Zippay and Hofmann 2010	Abalone	*Haliotis rufescens*	USA
Ellis et al. 2009	Periwinkle	*Littorina obtusata*	UK
3.2.7.2 / Bivalves			
Goethel et al. 2017	Clams	*Macoma calcarea*, *Astarte montagui*, *Astarte borealis*	Chukchi Sea
Bylenga et al. 2015	Clam	*Laternula elliptica*	Antarctica
Bylenga et al. 2017	Clam	*Laternula elliptica*	Antarctica
3.2.7.3 / Cephalopods			
Rosa et al. 2013	Cuttlefish	*Sepia officinalis*	Portugal
Rosa and Seibel 2008	Jumbo squid	*Dosidicus gigas*	USA
Hu et al. 2014b	Squid	*Sepioteuhis lessoniana*	Taiwan
Kaplan et al. 2013	Squid	*Doryteuthis pealeii*	USA
Sigwart et al. 2016	Cuttlefish	*Sepia officinalis*	France
Dorey et al. 2013	Cuttlefish	*Sepia officinalis*	Monaco
Gutowska et al. 2008	Cuttlefish	*Sepia officinalis*	France
Gutowska et al. 2010	Cuttlefish	*Sepia officinalis*	France
Spady et al. 2014	Squid	*Idiosepius pygmaeus*	Australia
3.2.8 / Echinoderms			
Bögner et al. 2014	Sea urchin	*Strongylocentrotus droebachiensis*	Svalbard
Ericson et al. 2010	Sea urchin, Nemertean worm	*Sterechinus neumayeri*, *Parborlasia corrugatus*	Antarctica
Ericson et al. 2012	Sea urchin	*Sterechinus neumayeri*	Antarctica
Yu et al. 2013	Sea urchin	*Sterechinus neumayeri*	Antarctica
Kapsenberg and Hofmann 2014	Sea urchin	*Sterechinus neumayeri*	Antarctica
Byrne et al. 2013	Sea urchin	*Sterechinus neumayeri*	Antarctica
Clark et al. 2009	Sea urchin	*Tripneustes gratilla*, *Pseudechinus huttoni*, *Evechinus chloroticus*, *Sterechinus neumayeri*	Antarctica, New Zealand, Cook Islands
Wood et al. 2011	Brittlestar	*Ophiocten sericeum*	Svalbard
Wood et al. 2010	Brittlestar	*Ophura ophiura*	UK

Ocean acidification treatment					
CO_2, μatm		CO_2, ppm		pH	
-	↓↑	-	↓↑	-	↓↑
				8.23	7.5, 6.5
~440	~1000-1080				
~400-450	~960-970, 1800-2000				
390	750, 1400				
371±160	944±888				
371±13	994±70				
		400	800, 1500, 2000, 3000		
		400	800, 1800		
		380	700-1000, >2000		
~450	~1000, 1600				
		380	570, 990		
				8.1	7.6
637	1267				
		~350	~560, 825		
~460	~710, 1040				
				8	7.5
				7.93±0.05	7.62±0.08
625±12	1286±253, 4134±169				
390	2200				
				~8	~7.8, 7.3
~390	~800, 1400				
		~650	~4000, 6000		
				8.01±0.04	7.10±0.03
447	626, 956				
~380	~180, 550-800, 1300, 2000				
528	1122, 2886, 5806				
		450	850, 1370		
410	510, 730				
400	650, 1000				
		433	927, 1417		
395-521	1119-1380				
		259±23	774±96, 1788±306		
		553±54	1400±75, 2546±205		

Table A3.1 cont.

Section / Source	Study species / community / assemblage		Location
Wood et al. 2008	Brittlestar	*Amphiura filiformis*	UK
Dupont et al. 2008	Brittlestar	*Ophiothrix fragilis*	Sweden
Chan et al. 2015	Sea urchin / Brittlestar	*Strongylocentrotus purpuratus / Amphiurafiliformis*	USA / Sweden
Hu et al. 2014a	Brittlestar	*Amphiura filiformis*	Sweden
Gonzalez-Bernat et al. 2013	Seastar	*Odontaster validus*	Antarctica
Dupont and Thorndyke 2012	Sea urchin, Seastar	*Strongylocentrotus droebachiensis, Leptasterias polaris*	Arctic
Verkaik et al. 2016	Sea cucumber	*Cucumaria frondosa*	Newfoundland
Yuan et al. 2016	Sea cucumber	*Apostichopus japonicus*	China
Morita et al. 2010	Coral, Sea cucumber	*Acropora digitifera*, *Holothuria* spp.	Japan
Yuan et al. 2015	Sea cucumber	*Apostichopus japonicus*	China
3.2.9 / Crustaceans			
Bailey et al. 2016	Copepod	*Calanus glacialis*	Svalbard
Bailey et al. 2017	Copepod	*Calanus glacialis*	Svalbard
Thor et al. 2016	Copepod	*Calanus glacialis*	Svalbard
Thor et al. 2018a	Copepod	*Calanus glacialis*	Svalbard (Kongsfjord / Billefjord) / West Greenland
Hildebrandt et al. 2014	Copepod	*Calanus glacialis*, *Calanus hyperboreus*	Fram Strait
Hildebrandt et al. 2016	Copepod	*Calanus finmarchicus*, *Calanus glacialis*	Fram Strait
Weydmann et al. 2012	Copepod	*Calanus glacialis*	Svalbard
Thor et al. 2018b	Copepod	*Calanus glacialis*	Svalbard
Niehoff et al. 2013	Mesozooplankton community		Svalbard
Engel et al. 2013	Plankton community		Svalbard
Walther et al. 2011	Spider crab	*Hyas araneus*	Germany, Svalbard
Schiffer et al. 2014	Spider crab	*Hyas araneus*	Sweden
Zittier et al. 2013	Spider crab	*Hyas araneus*	Svalbard
Long et al. 2013	Red king crab, Tanner crab	*Paralithodes camtschaticus, Chionoecetes bairdi*	Alaska
Appelhans et al. 2012	Seastar, Green crab	*Asterias rubens*, *Carcinus maenas*	Baltic Sea
Fehsenfeld and Weihrauch 2013	Green crab	*Carcinus maenas*	Canada
Fehsenfeld et al. 2011	Green crab	*Carcinus maenas*	Baltic Sea
Hammer et al. 2012	Green crab	*Carcinus maenas*	Norway
Arnold et al. 2009	European lobster	*Homarus gammarus*	UK
Small et al. 2016	European lobster	*Homarus gammarus*	UK
Agnalt et al. 2013	European lobster	*Homarus gammarus*	Norway
Bechmann et al. 2011	Shrimp / Mussel	*Pandalus borealis / Mytilus edulis*	Norway

Ocean acidification treatment					
CO_2, µatm		CO_2, ppm		pH	
-	↓↑	-	↓↑	-	↓↑
				8	7.7, 7.3, 6.8
				8.1	7.9, 7.7
458±32 / 425±13	1078±48, 2993±188 / 1126±83				
				8.1	7.6, 7.3, 7.0
327	691, 1130, 4604				
350	1275				
446±22	1427±100				
		~380	~750, 1900		
		400-475	775-1005, 930-1260, 905-1660, 2115-3585, 12600-21100		
601±10	962±15, 1441±21, 2801±25				
530	320, 800, 1700				
530	320, 800, 1700				
335-361	871-1060				
450±95 / 446±93 / 436±64	712±134 to 18567±2163 (8 treatments) / 638±49 to 4526±499 (6 treatments) / 721±91 to 19456±3521 (8 treatments)				
390	3000				
390	1120, 3000				
				8.2	7.6, 6.9
				~8.0	~7.5
185	270, 375, 480, 685, 820, 1050, 1420				
178	180, 255, 345, 435, 611, 701, 892, 1136				
		380	710, 3000		
450	3300				
380	750, 1120, 3000				
438±9	792±7, 1638±14				
650	1250, 3500				
				7.7	7
				8.00-8.12	7.24-7.36
~490	~ 2600, 7600, 16000, 30000				
		380	1200		
450	1100, 9000				
~690	750, 1200				
368-361 / 419-469	1291-1332 / 1388-1493				

Table A3.1 cont.

Section / Source	Study species / community / assemblage		Location
Arnberg et al. 2013	Shrimp	*Pandalus borealis*	Norway
Dupont et al. 2014	Shrimp	*Pandalus borealis*	Sweden
Findlay et al. 2010	Barnacle	*Semibalanus balanoides*	Svalbard
3.2.10 / Other invertebrates			
Lesniowski et al. 2015	Scyphozoan jellyfish	*Cyanea capillata*, *Chrysaora hysoscella*	Germany
Cross et al. 2015	Brachiopod	*Liothyrella uva*	Antarctica
McClintock et al. 2009	Bivalves, Limpet, Brachiopod	*Laternula elliptica*, *Yoldia eightsi*, *Nacella concinna*, *Liothyrella uva*	Antarctica
Cross et al. 2016	Brachiopod	*Calloria inconspicua*	New Zealand
Turner et al. 2015	Polychaete	*Sabella spallanzanii*	Mediterranean
Verkaik et al. 2017	Polychaete	*Ophryotrocha* sp.	Canada
Lee et al. 2017	Meiofaunal assemblage		Chile
Dashfield et al. 2008	Nematode community		Norway
Hale et al. 2011	Benthic community		UK
Meadows et al. 2015	Meiobenthic community		UK
Widdicombe et al. 2009	Macrofaunal and nematode assemblage		Norway
3.2.11 / Fishes			
Kunz et al. 2016	Polar cod / Atlantic cod	*Boreogadus saida* / *Gadus morhua*	Norway
Stapp et al. 2015	Atlantic cod	*Gadus morhua*	Germany
Hu et al. 2016	Atlantic cod	*Gadus morhua*	Sweden
Michael et al. 2016	Atlantic cod	*Gadus morhua*	Sweden
Leo et al. 2017	Polar cod, Atlantic cod	*Boreogadus saida*, *Gadus morhua*	Svalbard
Frommel et al. 2010	Baltic cod	*Gadus morhua*	Baltic Sea
Dahlke et al. 2017	Atlantic cod	*Gadus morhua*	Sweden
Stiasny et al. 2016	Atlantic cod	*Gadus morhua*	Western Baltic Sea / Barents Sea
Maneja et al. 2013b	Atlantic cod	*Gadus morhua*	Norway
Frommel et al. 2012	Atlantic cod	*Gadus morhua*	Norway
Frommel et al. 2014	Atlantic herring	*Clupea harengus*	Norway
Frommel et al. 2013	Baltic cod	*Gadus morhua*	Baltic Sea
Schmidt et al. 2017a	Polar cod, Atlantic cod	*Boreogadus saida*, *Gadus morhua*	Svalbard
Schmidt et al. 2017b	Polar cod, Atlantic cod	*Boreogadus saida*, *Gadus morhua*	Norway
Melzner et al. 2009	Atlantic cod	*Gadus morhua*	Norway / Germany
Maneja et al. 2013a	Atlantic cod	*Gadus morhua*	Norway
Jutfelt and Hedgärde 2013	Atlantic cod	*Gadus morhua*	Sweden
Jutfelt and Hedgärde 2015	Atlantic cod	*Gadus morhua*	Sweden
3.2.12 / Seabirds and mammals			
3.3 / Ecosystems, habitats			
Tarling et al. 2016	Pelagic community		Nordic, Scotia, Weddell Seas
Bibby et al. 2007	Gastropod	*Littorina littorea*	UK

Ocean acidification treatment					
CO_2, μatm		CO_2, ppm		pH	
-	↓↑	-	↓↑	-	↓↑
337-474	1038-1437				
459±5	1368±7				
352±28	1086±95, 2429±335				
		200	800		
365±67	725±133, 1221±179				
				8.2	7.4
465±83	1130±12, 1536±235				
502±10	6144±553				
		475±68	1508±216		
		400	1000		
				8	7.5
				8	7.7, 7.3, 6.7,
				8	7.7, 7.3, 6.7
				8	7.3, 6.5, 5.6
390	1170				
				7.9	7
				8.1	7.8, 7.6
550	1200, 2200				
400	1170				
				~8.1	~ 7.6
400	1100				
~400	~1100				
370	1800, 4200				
380	1800, 4200				
				8.08	7.45, 7.07
380	560, 860, 1120, 1400, 4000				
374-515	852-1416				
396-548	915-1272				
				8.01±0.08 / 8.02±0.06	7.01±0.03 / 7.30±0.03
370	1800, 4200				
550	1170				
500	1000				
300	750, 1000				
				7.97-8.02	6.56-6.73

Table A3.1 cont.

Section / Source	Study species / community / assemblage		Location
Lesniowski et al. 2015	Scyphozoan jellyfish	*Cyanea capillata*, *Chrysaora hysoscella*	German Bight
Goethel et al. 2017	Bivalves	*Macoma calcarea*, *Astarte montagui*, *Astarte borealis*	Chukchi Sea
3.4 / Acclimation and adaptation			
Form and Riebesell 2012	Coral	*Lophelia pertusa*	Norway
Leo et al. 2017	Polar cod, Atlantic cod	*Boreogadus saida*, *Gadus morhua*	Svalbard
Dupont and Thorndyke 2012	Sea urchin, Seastar	*Strongylocentrotus droebachiensis*, *Leptasterias polaris*	Arctic
Monier et al. 2014	Microbial community		Ellef Ringnes Island
Thabet et al. 2017	Pteropod	*Clione limacina*	USA
Ragazzola et al. 2013	Coralline red algae	*Lithothamnion glaciale*	Kattegat
Lohbeck et al. 2012	Coccolithophore	*Emiliania huxleyi*	Norway
Schaum et al. 2013	Picoplankton	*Ostreococcus*	Thau lagoon, Mediterranean, North Sea, Atlantic Ocean, Red Sea, English Channel, North Sea, Spanish coast, West Mediterranean
Thor and Dupont 2015	Copepod	*Pseudocalanus acuspes*	Sweden
De Wit et al. 2016	Copepod	*Pseudocalanus acuspes*	Sweden
Miller et al. 2012	Anemonefish	*Amphiprion melanopus*	Australia
3.5 / Interactive effects, multistressor environment			
Coello-Camba et al. 2014	Phytoplankton community		Arctic Ocean
Gordillo et al. 2016	Chlorophyte, Rhodophytes, Phaeophytes	*Monostroma arcticum*, *Phycodrys rubens*, *Ptilota plumosa*, *Alaria esculenta*, *Desmarestia aculeata*, *Saccorhiza dermatodea*	Svalbard
Iñiguez et al. 2016	Kelp	*Saccharina latissima*, *Laminaria solidungula*	Svalbard
Olischläger et al. 2017	Kelp	*Saccharina latissima*	Spitsbergen, North Sea
Lischka and Riebesell 2012	Pteropod	*Limacina helicinaia*, *Limacina retroversa*	Svalbard
Dahlke et al. 2017	Atlantic cod	*Gadus morhua*	Sweden
Leo et al. 2017	Polar cod, Atlantic cod	*Boreogadus saida*, *Gadus morhua*	Svalbard
Hoppe et al. 2018	Phytoplankton community		Baffin Bay
Gordillo et al. 2015	Kelp	*Alaria esculenta*, *Saccharina latissima*	Svalbard
Marques et al. 2017	Foraminifera	*Amphistegina gibbosa*	Brazil
Maat et al. 2014	Picoeukaryote	*Micromonas pusilla*	North Sea
Reymond et al. 2013	Foraminifera	*Marginopora rossi*	Australia
Haynert and Schönfeld 2014	Foraminifera	*Ammonia aomoriensis*	Baltic Sea
Celis-Plá et al. 2015	Macroalgal assemblage		Italy
Lesniowski et al. 2015	Scyphozoan jellyfish	*Cyanea capillata*, *Chrysaora hysoscella*	German Bight
Maier et al. 2016	Coral	*Madrepora oculata*	Adriatic Sea
Dashfield et al. 2008	Nematode community		Norway
Falkenberg et al. 2012	Kelp, Turf	*Ecklonia radiata*, *Feldmannia* spp.	Australia
Falkenberg et al. 2014	Turf, Gastropod	*Feldmannia* spp., *Austrocochlea concamerata*, *Austrocochlea odontis*	Australia

Ocean acidification treatment					
CO$_2$, µatm		CO$_2$, ppm		pH	
-	↓↑	-	↓↑	-	↓↑
		200	800		
637	1267				
~509	~605, 856, 981				
400	1170				
350	1275				
		400	880, 1800		
~440	~1000-1080				
408	566, 770, 1024				
400	1100, 2200				
		380	1000		
400	900, 1550				
400	900, 1550				
430	581, 1032				
		380	1000		
		390	1100		
		390	1200		
380	800, 1500				
350	650, 880				
400	1100				
400	1170				
380	1000				
		380	1000		
				8.1	7.8, 7.5, 7.2
370	750				
		380	700, 1000		
566	1195, 2108, 3843				
500	700-800, 1200				
		200	800		
		400	800, 1600, 2000		
				8	7.5
		280-380	550-650		
		~380	~580		

4. Socio-economic impacts of Arctic Ocean acidification on fisheries

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4.1 Introduction

Arctic and subarctic fisheries are important and valuable, and much has been done to keep them that way. Commercial fisheries in Arctic and subarctic water account for a tenth of the world's catch (AMAP, 2013; CAFF, 2013). Subsistence fisheries along coasts and up the rivers in which anadromous fish swim provide extensive nutritional and cultural benefits to Arctic residents and are based on the deep knowledge and extensive experience of Indigenous peoples and others who live in the Arctic (AMAP, 2013; CAFF, 2013). In addition to effective fisheries management practices by many Arctic countries in their northern waters, several conservation measures have been put in place in recent years to protect Arctic marine ecosystems such that they can continue to provide these benefits for humans, now and into the future. These measures include the *U.S. Arctic Fisheries Management Plan* (NPFMC, 2009), limiting commercial harvests in U.S. Arctic waters until there is sufficient scientific knowledge to support sustainable fisheries; the *Beaufort Sea Integrated Fisheries Management Framework*, establishing a similar system in the Canadian portion of the Beaufort Sea (Ayles et al., 2016); and the 2017 international *Agreement to Prevent Unregulated High Seas Fisheries in the Central Arctic Ocean*, which extends the same logic to the waters beyond any country's exclusive economic zone (EEZ; Hoag, 2017).

Ocean acidification (expressed by the decrease in pH in ocean waters in contact with an atmosphere of increasing carbon dioxide) has the potential to undermine both the fisheries and the management efforts to make those fisheries substantial and sustainable (Cooley and Doney, 2009). Chapter 3 described the various ways that increasingly acidic seawater can affect marine organisms and the interactions among them. Ocean acidification can affect a species directly, by altering its growth, development, or behavior at one or more life stages. Ocean acidification can also affect species indirectly, by altering foodweb and predator-prey relationships. It is important to note that not all effects are necessarily negative, and also that ocean acidification and its effects will not occur in isolation but in combination with other stressors such as ocean warming and human activity. The magnitude of changes anticipated, either positive or negative, prompt that section of the report to conclude that forecasted ocean acidification "is likely to be sufficient to drive changes in Arctic organisms and ecosystems to an extent that will impact the associated human societies".

With that background, this chapter draws on five case studies prepared as part of this assessment and included at the end of the report as annexes (Figure 4.1, Table 4.1). The five case studies examine the likely effects of Arctic Ocean acidification on marine organisms and their fisheries, highlighting the socio-economic effects that are the focus of this chapter. Three case studies look at specific fisheries: Norwegian kelp and sea urchins (Annex 2), Barents Sea cod (Annex 3), and Greenland shrimp (Annex 4). The other two consider regional effects and responses to the fishing sector in Alaska (Annex 5) and in the western Canadian Arctic (Annex 6). While the case studies do not encompass all of the fisheries in the Arctic, they nonetheless provide compelling examples of what may be expected as the Arctic Ocean becomes more acidic, and ideas for addressing and taking adaptive measures now in anticipation of projected effects. A conundrum for decision-makers on how to manage these fisheries, however, is that while most of the effects will not become apparent for some time to come, inaction now could mean fewer choices and worse outcomes in the decades ahead. In other words, future well-being depends on today's response.

4.2 Expected effects of ocean acidification on selected Arctic fisheries

This section provides an overview of each of the five case studies, with particular emphasis on local and regional socio-economic effects, such as employment, income, and community well-being. More details, including supporting references, are available in the case studies themselves.

4.2.1 Norwegian kelp and sea urchins

Sea urchins are in high demand globally. As supplies dwindle, prices are expected to increase along with the harvest area. In northern Norway, there is a large and thus far commercially unexploited stock of green sea urchin (*Strongylocentrotus droebachiensis*). From another perspective, urchins may be regarded as a pest that consumes valuable kelp forests through overgrazing, so that in some areas there may be interest in culling sea urchins to allow the kelp forest to recover. Kelp (notably sugar kelp *Saccharina latissimi*) has been harvested commercially in Norway for more than 50 years, and kelp forests provide important habitat for many commercially valuable fish species, among other benefits. A key question is how ocean acidification will affect sea urchins in the region and the kelp they graze on. From the perspective of urchin harvesting, a second key question is how fishery managers can adjust rules to achieve sustainable utilization and maximal value of urchin harvests. For this case study, researchers created a model to investigate optimal minimum size limits for harvested sea urchins to maximize the yield of present-day and future harvests.

Based on results from experimental and field data, the modelled urchin populations were strongly affected by near-term (30-year) warming and acidification, due to sensitivity in the larval and juvenile stages, while kelp regrowth was weakly enhanced. The resulting model simulations suggest that harvest yield of sea urchins would be strongly reduced (roughly seven-fold) in the future scenario, partly due to acidification, but mainly due to warming. A minimum harvested size of around 5 cm was found to give the highest sustainable harvest yield (in terms of gonad biomass) in both present-day and future scenarios. These results, however, should be treated with caution because

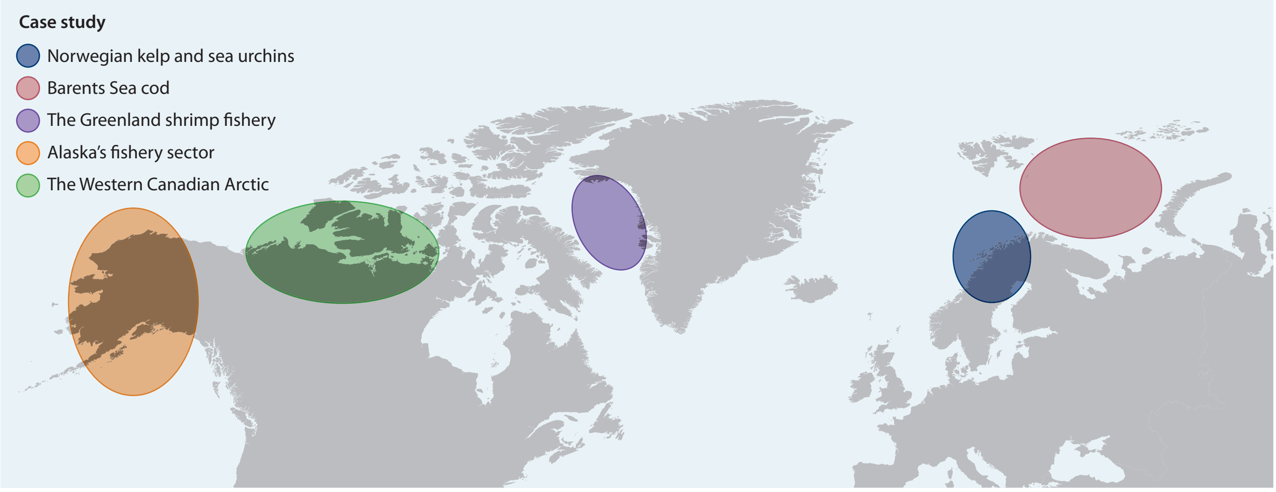

Figure 4.1 Locations of the case studies.

they depend on estimates of urchin life stage sensitivities that are currently not well constrained by experimental data. More experiments are needed at the required moderate levels of acidification and warming, and more field investigations are needed to understand the north-to-south variability in urchin abundance along the Norwegian coast. Nevertheless, this case study does suggest that ocean warming and acidification may have strong impacts on the size of the harvestable sea urchin stock in northern Norway, even within the next 30 years.

4.2.2 Barents Sea cod

In the northeast Atlantic Ocean, cod fishing has been a major economic activity for centuries. Both ocean warming and ocean acidification are expected to have negative effects on Barents Sea cod (*Gadus morhua*), known as the Northeast Arctic cod stock. Ocean acidification, according to laboratory experiments, will increase mortality during the larval stage of cod. Ocean warming will reduce recruitment in this population. This case study considers the economic effects of the combination of warming and acidification if carbon dioxide emissions continue through this century in a 'business as usual' scenario. Arctic seas, including the Barents Sea, are expected to be especially vulnerable to both ocean warming (because the Arctic is warming faster than the global average) and ocean acidification (due to undersaturation of aragonite – one of the two most common naturally occurring forms of calcium carbonate – in Arctic surface waters).

The Northeast Arctic cod stock spawns along the Norwegian coast, migrating northwards over spring and summer into the Barents Sea. In recent years, the distribution of cod has shifted northwards, as shown by cod catches. Warming of the Barents Sea to date has been beneficial to the cod, responsible in part for catches returning to levels not seen since the mid-1970s, when they were about 900,000 tonnes per year. Further warming, however, is expected to have negative effects as described above.

The model used in this case study examines the combined effects of fishing, warming, and acidification, with the ability to vary each of these factors independently. The model indicates that acidification greatly increases the risk of a stock collapse. As a result, under acidifying conditions, catch levels must be much lower to be sustainable. Warming is also shown by the model to have a similar effect, but not as pronounced, when waters are above an optimal temperature for cod of about 4.5°C. The results were then considered in light of fishery management. At the optimal temperature, and without ocean acidification, a sustainable catch of 900,000 tonnes per year is possible. This catch level corresponds to net annual revenues of 2.3 billion Norwegian kroner (about USD 285 million). A high, lucrative catch goes hand in hand with high employment in the fishery, as harvests at that level are only possible with enough fishers at sea. Catches and revenues decrease sharply either side of the temperature optimum, with revenue reaching zero at about 6.5°C.

When ocean acidification is added to the model, prudent fishery management requires keeping the spawning stock biomass higher to account for the effects of acidification. Combined with the effects of acidification itself, the result is an optimal catch below 150,000 tonnes per year, with a net revenue under 300 million Norwegian kroner (about USD 37 million). The number of jobs supported by the fishing sector would also be much lower. When combined with acidification, the effect of temperature is more pronounced, with catch and revenue reaching zero at 6°C instead of 6.5°C. Ocean acidification has no effect on the optimal temperature for the cod stock and fishery, which remains 4.5°C. In short, ocean acidification under the 'business as usual' emissions scenario has the potential to turn a large fishery employing many workers into a much smaller fishery with fewer jobs and far less revenue.

4.2.3 The Greenland shrimp fishery

The West Greenland shrimp (*Pandalus borealis*) fishery is a vital component of Greenland's overall income and a major source of revenue and employment for many communities. Unfortunately, the potential and likely effects of ocean acidification on the shrimp stocks in the Baffin Bay / Davis Strait region are not known with any certainty. There is some evidence that the physiology of northern shrimp may be relatively tolerant of acidification. That is, they appear able to offset the effects of ocean acidification on the acidosis of body fluids. Calcification in other shrimp species has been found to be not greatly affected by anticipated levels of acidification for the world's oceans. Some effects of acidification may be worsened in warmer waters, although warmer waters may also counteract some

Table 4.1 Summary of the case studies, key findings, and management implications. Note all key findings reported here are for more than two decades from now. Few shorter-term effects were identified, although confidence is low due to high uncertainty.

Region: Topic	Key findings	Management implications
Norway: Sea urchins and kelp	Ocean acidification and warming reduce potential urchin harvest by a factor of seven	A minimum size limit of about 5 cm for harvested urchins is effective regardless of population size
Norway: Barents Sea cod	Ocean acidification makes stock collapse more likely, especially in combination with warming	Catch limits need to be more conservative to leave a buffer for the effects of ocean acidification and warming
Greenland: Shrimp	Insufficient data for a bioeconomic model	Management flexibility and economic diversification can help deal with consequences of reduced shrimp catches
Alaska: King crab	Decrease in crab catch due to ocean acidification	Attention to adaptive capacity, especially for rural communities
Canada: Western Arctic Bioregion	Increased marine productivity due to warming, with minor effect from ocean acidification	Attention to foodweb changes and implications for Indigenous communities and equity in any fisheries

of the negative effects of acidified waters. Indirect effects are also possible. For example, if shrimp develop more slowly in more acidic waters, they may be vulnerable to predation for a longer period. Another concern is the effect of acidification on the taste of shrimp to humans, as one component of their economic value. For example, when shrimp were exposed to more acidic waters for three weeks, they were rated lower for appearance and taste by human consumers.

The shrimp fishery in Greenland's waters includes both an offshore and a coastal fleet, with the total harvest set as a total allowable catch (TAC). The TAC is allocated among holders of individual transferable quotas (ITQs), which are divided between the offshore and coastal fleets and with a portion of the TAC sold to European Union fishers. Current management of the shrimp fishery, based on stock assessment models, appears to be robust and conservative. For example, shrimp biomass is currently about 11% above the level associated with maximum sustainable yield, which is regarded as a sensible buffer against unforeseen changes despite the fact that it also means current harvests are lower than their potential.

The catch by the coastal fleet is processed entirely in Greenland and at least 25% of the offshore catch must also be processed onshore in Greenland. Altogether, the shrimp fishery has accounted for 32–44% of the total value of Greenland's fisheries in the past five years. The fishing sector as a whole is responsible for 95% of Greenland's exports (in 2017) and 20% of its gross domestic product (in 2016). The total shrimp catch has decreased since 2012, although its value has remained steady. The lower catch, however, has led to reductions in onshore processing facilities, with large effects on local employment and economies in the communities where facilities have closed. While the role of ecological factors in changes to the fishery remain unclear, it is certain that the shrimp fishery is greatly affected by changing economic factors over time, such as global demand. The effects of ocean acidification will occur within a greater bioeconomic context of changing markets and economics and changing oceanographic conditions.

Modeling of the bioeconomics of Greenland's shrimp fishery is hampered by the high levels of uncertainty at all stages of analysis, from the effects of ocean acidification on shrimp to the environmental and societal factors that influence the way the fishery is conducted. An attempt to model the bioeconomics resulted in the conclusion that, with currently available data, this approach to assess the effects of ocean acidification on the Greenland shrimp fishery is futile. But there are other approaches that may be helpful. A simpler model that does not attempt to combine all factors into a single analysis but focuses instead on the shrimp population may yield useful insights.

The current management approach has a high level of adaptability, takes an ecosystem approach, and includes consideration of socio-economic concerns, all of which make for a well-managed fishery. If it is not currently possible to create a bioeconomic model to predict the effects of ocean acidification it is nonetheless possible to identify management options despite uncertainty. For the Greenland shrimp fishery this includes three specific suggestions:

- Monitor the spatial extent of the shrimp fishery to gain insight into the various factors of change. Such information can help with pro-active adaptation measures. Canada also fishes northern shrimp on its side of the maritime boundary in the Baffin Bay / Davis Strait region. Better cooperation between Greenland and Canada on monitoring for ecological change and for planning how to share changes in spatial stock dynamics would assist in building greater resilience.
- Increase economic resilience, especially via diversification throughout Greenland. This can include investing in facilities that can process many different species to reduce dependence on a specific fishery, and also in education and job training outside of fisheries.
- Invest in cost-effective research on specific concerns related to ocean acidification, such as its effects on the taste of shrimp and on shrimp reproduction and genetics. Such research would be substantially easier and cheaper than extensive oceanographic fieldwork and is likely to produce useful results more quickly and can help focus ecological research on questions most likely to improve management of the fishery.

The case of the West Greenland shrimp fishery concerns a single species caught in the waters of a single country and sold primarily in a well-functioning global export market. The fact that such a case defies the creation of a reliable bioeconomic model suggests that this is also an unrealistic goal for other locations, especially those with more complex harvests and harvest interactions. This finding does not, however, suggest that nothing can be done. Instead, attention to the overall robustness of the fishery and the fishery management system can have many positive results, including being better prepared for ocean acidification.

4.2.4 Alaska's fishery sector

Alaska's fishery sector includes major commercial fisheries, culturally vital subsistence fisheries, and popular local and tourist-driven recreational fisheries. Ocean acidification has the potential to disrupt harvests, with effects on employment and income, cultural continuity and health, community demographics, and individual well-being. This case study examined risk in terms of patterns of human dependence on marine resources across all of Alaska. It then used the Bristol Bay red king crab (*Paralithodes camtschaticus*) fishery as a specific example for the development of a bioeconomic model to evaluate how ocean acidification may affect maximum sustainable yield and maximum economic yield for that fishery.

Risk has three components: hazard, exposure, and vulnerability. These components are combined to assign a risk index for different regions of Alaska. The hazard posed by ocean acidification to Alaska's fishing sector was quantified using surface pH and saturation states of aragonite and calcite. Exposure was considered separately for shellfish, salmon, and other finfish, in terms of both nutritional and commercial value of the harvests. Vulnerability considered sensitivity (i.e., the degree of human reliance on a given species) and adaptive capacity (i.e., the ability of humans to respond or adapt effectively).

The risk index shows that southern Alaska faces the greatest risk from ocean acidification, due to high dependence on susceptible species for both food and income, combined with the prediction of rapid changes in ocean chemistry. Adaptive capacity in this region is generally good, although not in rural areas with low job diversity, employment, and education as well as a high cost of living. This lack of adaptive capacity is high throughout the state's rural areas, although large and diverse subsistence harvests in northern and interior Alaska help reduce sensitivity.

Identifying the areas of the social-ecological system that are most vulnerable can help develop localized policies to reduce risk. Such policies are best developed at the community level, although state and federal governments can provide support, as can scientists and others. A closer look at the Bristol Bay red king crab fishery in southwestern Alaska helps illustrate this point. The fishery is managed through a quota system in which quota owners can lease their shares to vessel owners, thereby creating two income streams and wide-ranging social effects in the region and beyond.

The bioeconomic model for this fishery included a pre-recruit component for the crabs, a post-recruit component including the effects of the fishery, and a profit component for the commercial fishing sector. The results show a decline in catches over the next 50 to 100 years under two scenarios of an acidifying ocean. Profits also declined but not nearly as much as the catches due to an anticipated increase in the price of the crabs. Overall societal losses from ocean acidification, such as indirect economic benefits as well as equitable participation in fisheries, were found to be greater than changes to profits. Together, these findings suggest that the magnitude of the socio-economic effects of ocean acidification will depend in large part on world market demand, because reduced catches may be partially or completely offset by higher prices or stable catches could be undermined by falling prices. These effects are likely to be limited in the next decade or two but economic losses may eventually total tens of millions of U.S. dollars per year, a few percent of the economic value of Alaska's crab fisheries (McDowell Group, 2015). Indirect costs such as losses in the sectors that support fisheries could be even higher, because commercial fishing is an important sector in Alaska.

4.2.5 The Western Canadian Arctic

The Western Arctic Bioregion in Canada comprises the Inuvialuit Settlement Region of the Northwest Territories and Yukon and the Kitikmeot Region of Nunavut. Several Western Canadian Inuit communities are located in this area, whose residents continue their traditional ways of harvesting marine species and using them for food, clothing, arts, and other purposes. Social and economic changes have altered diets, as has awareness of persistent pollutants in many traditional foods. Climate change is affecting hunting and fishing practices, both through physical changes to the environment and effects on the foodweb and the distribution of animals. At present, there are no offshore commercial fisheries here due to the *Beaufort Sea Integrated Fisheries Management Framework* mentioned in Section 4.1, but that policy could change as more is learned about the fish stocks and their ecosystem.

Assessing the effects of ocean acidification in the western Canadian Arctic, as elsewhere, is hampered by lack of data, especially concerning physiological effects on the species of interest. A key species in the marine ecosystem of the region is Arctic cod (*Boreogadus saida*), the distribution of which has already shifted northwards. Even so, little is known about the ways in which Arctic cod will respond to stressors, much less to the combination of multiple stressors acting simultaneously. Despite these uncertainties a model was created to examine the effects of climate change and ocean acidification on the distribution and abundance of Arctic cod. In addition to the ecological model, an economic model used the ecological model's results to explore the potential scale of commercial fisheries in the region and their value now and in the future.

Ocean acidification is expected to be particularly strong in polar regions, with surface waters in the Arctic Ocean becoming more acidified and reduced in calcium carbonate saturation over the next century. Ecological changes in the region's marine environment over the past 45 years have had the net effect of increasing biomass for most components of the foodweb. Fish biomass increased by 59%, with indications that previously sparse species such as capelin (*Mallotus villosus*) could become far more abundant. This trend is expected to continue as the sea in this region continues to warm and undergo further ecosystem transitions, which would be to the detriment of Arctic cod and its predators unless those predators could switch to other prey. Using simplified parameterizations, ocean acidification appears to have only a modest effect, causing slightly lower increases in the abundance of most finfish species compared to the effect of climate change alone. Ocean acidification effects on invertebrates are likely to be higher.

Economically, if commercial fisheries existed in the western Canadian Arctic at present, the total catch is estimated to be over 55,000 tonnes per year, valued at over USD 52 million, although the distribution of the economic benefit locally and nationally is unclear. Finfish would account for 90% of the mass and invertebrates would make up 35% of value, since they fetch higher prices on average. The ecological model shows an increase in catch potential due to the poleward shift in the distribution of many fish species by 2100, potentially offsetting the decreases in Arctic cod. If the carbon emission targets proposed in the Paris Agreement are met, the catch and value would increase by roughly 20%. Under a scenario with higher carbon dioxide emissions the catch could double in volume and triple in value over the same period. Ocean acidification would primarily affect the invertebrate harvest, but the reduction would amount to about 1% of the volume and value of the fisheries in the region.

Culturally, the outcomes are not as clear. The decrease in Arctic cod abundance could affect its predators, including highly valued marine mammals such as ringed seals (*Pusa hispida*) and beluga (*Delphinapterus leucas*). An overall increase in marine productivity could help offset the loss of Arctic cod. Environmental changes since the 1980s suggest a mixed result with ringed seals, beluga, and chicks of black guillemot (*Cepphus grylle*) seeing declines in body condition, growth, or production. Subadult bowhead whales (*Balaena mysticetus*) and Arctic char (*Salvelinus alpinus*), however, both showed improved body condition, likely to be due to changes in sea ice. The long-term effects remain unclear. Ocean acidification appears to play a lesser role than other drivers of change in the region, but potentially synergistic effects have not yet been taken into account.

4.3 Ocean acidification, Arctic fisheries, and other factors

As the five case studies illustrate, the effects of ocean acidification are neither uniform nor can they be reliably predicted. Acidification will not be uniform throughout the ocean, different species will be affected in different ways and to different degrees, ecosystem dynamics vary, and the individuals and communities that rely on marine resources do so in different ways and to different extents. All of these factors vary over time and space and all have elements of complexity and uncertainty. Furthermore, ocean acidification is not occurring in isolation, but alongside extensive social, economic, and ecological changes affecting fisheries. Climate change, largely driven by the same rising carbon dioxide levels that cause ocean acidification, is already having extensive effects on the world's oceans, not least in the Arctic where warming is amplified and seasonal sea ice change is one of the most visible signs of global warming (Serreze et al., 2009).

All five case studies emphasize the high degree of uncertainty concerning how more acidic waters will affect species directly. The divergent results of the models and other analyses – stemming perhaps from differences in methodology as well as different ecosystem dynamics in different regions and for different species – are one example of that uncertainty, which is exacerbated when foodweb dynamics, habitats, and other ecosystem considerations are included. The Western Canadian Arctic case study found that ocean acidification is likely to play a minor role (Annex 6), whereas nearby Alaska may eventually see large declines in the value of lucrative fisheries such as that for the Bristol Bay red king crab (Annex 5). From the case studies, there appears to be little evidence that ocean acidification will have major effects on Arctic fisheries and societies in the next decade or two. The longer-term prognosis is less clear. There is reason to think that ocean acidification will not be a major disrupter, but there is also sufficient uncertainty in the case studies that such an outcome cannot be ruled out. In brief, the prudent course is to continue to evaluate the effects of ocean acidification and to develop response strategies that are robust over a range of ecosystem conditions.

Climate change appears to be the dominant stressor causing changes in fish distributions and abundance worldwide. In the western Canadian Arctic, climate change may lead to increased productivity and abundance of many species, creating an increase in the volume and value of potential commercial fisheries in the region (Annex 6). In Norway, climate change and ocean acidification are likely to combine over time to greatly decrease the sea urchin harvest (Annex 2) and to greatly increase the likelihood of collapse of the Northeast Arctic cod stock (Annex 3). Again, the prudent course is to manage the fisheries with the unwelcome changes in mind, to reduce the chances of major failures resulting from unanticipated effects on fish stocks and ecosystems. Such a course, however, may mean reducing harvests today, as is the choice that has been made for the West Greenland shrimp fishery (Annex 4).

The economic models and the social analyses in the case studies show how important socio-economic trends will be in determining the future value of Arctic fisheries and their cultural and nutritional significance. Changes in prices, driven by the state of global markets, will have profound effects on the value of Arctic fisheries, regardless of their volume. The prospect of a sea urchin fishery in northern Norway is one example of an opportunity created by global demand (Annex 2). Markets can also work in the opposite direction, as can be seen in the decline in price for wild Pacific salmon, due in large part to competition from farmed Atlantic salmon as a cheaper substitute in the marketplace (Knapp et al., 2007). How markets will be affected by changes in fish distributions and abundance worldwide remains to be seen. The growth of aquaculture may also spur increased demand for fishes that are currently in low demand for human consumption (Hasan and Halwart, 2009), perhaps someday to include Arctic cod. Subsistence uses of fish are changing as a result of environmental change as well as social change (Fall et al., 2013). Managing fisheries and ecosystems prudently, keeping options open so that Arctic peoples can make their own choices, is a worthy goal.

4.4 Understanding and action under uncertainty

Uncertainty is a recurring theme in all five case studies, as has been found in other attempts to examine socio-economic effects of ocean acidification, for example in the Mediterranean

(Rodrigues et al., 2013). Observational data are in many cases insufficient for all parts of the system. There is also considerable uncertainty regarding many of the assumptions made in the models, and it is unclear if those assumptions include all the factors that may affect the eventual outcomes of ocean acidification, especially in combination with other changes in the ocean. The West Greenland shrimp fishery case study is explicit in its conclusion that a bioeconomic model is too unreliable to be of any use in determining management approaches (Annex 4). Models in the other case studies were carried through to producing results but those results were all presented with extensive cautions about the ways in which they could be wrong.

This is not to say that nothing is known about Arctic Ocean acidification and its effects. There is ample reason to continue to pay close attention to changing acidity and calcium carbonate saturation levels as well as biological effects, and further research can help improve the models. Other chapters in this report have addressed those topics. Continued change in the waters of the Arctic will also provide further insight into the effects of ocean acidification and the ways in which it interacts with other stressors, provided it is monitored in a way that allows causes and effects to be identified. Changes to foodwebs may occur on longer time scales than changes in water chemistry, complicating the ability to connect causes and effects. In the meantime action is necessary. While the more extensive effects of ocean acidification are likely to be some time off, many aspects of managing and conducting fisheries will also take time to change.

The robustness of management approaches to varying levels of effects from climate change and ocean acidification can be evaluated to determine whether current strategies will continue to be prudent and effective both for today's fisheries and for those of the future. Investments in fisheries-related infrastructure and human capacity can also be made with flexibility and adaptation in mind. Training a large number of fishers and investing in species-specific vessels and processing plants may lead to major economic and social disruptions if the species and fisheries decline. A diversified economy, by contrast, can have many benefits, not least providing resilience across a range of future scenarios. Further research into the social and economic aspects of resilience to ocean acidification and warming could address these and related topics to provide greater insight into what is possible and what is likely to be effective. As has been noted with regard to climate change (Schneider and Kuntz-Duriseti, 2002) much can be done despite uncertainty.

4.5 Mitigation and adaptation options

Ultimately, ocean acidification and climate change can only be mitigated by global action. In the Arctic and within Arctic countries, there are a number of locally based options for reducing some of the effects and for adapting to those that cannot be prevented. As noted in the previous section, fisheries management that takes ocean acidification into account can provide a buffer against some of the effects that may occur. Ecosystem-based management can also help sustain overall ecosystem health (e.g., Pikitch et al., 2004), which is especially important where several species are harvested. The Norwegian kelp and sea urchin study is one such example, where the harvests of two resources can affect one another (Annex 2). In the Western Canadian Arctic case study, a recognition that new species may provide new opportunities is important, as is continued monitoring to see how a decline in Arctic cod may affect its predators, which are important components of local diets and economies (Annex 6). If the predators can switch to other prey, these species may persist. If they are unable to find prey an overall increase in commercially valuable catches may still cause large disruptions to traditional human practices and well-being.

Adaptation options also include a range of possibilities on the economic and social fronts. Specialization may be an effective strategy under stable or at least predictable conditions, but generalists may fare better in the face of change and uncertainty. If the Norwegian fishing sector remains focused on cod, or the West Greenland fishery on shrimp, a change in the foodweb structure of those seas could lead to high unemployment and loss of incomes. It is not clear that a replacement catch could be found in either case but preparing for declines in the main target species can include identifying other fishing opportunities, such as sea urchins in northern Norway (Annex 2), as well as economic diversification beyond the fishing sector (Annex 4). When considering how best to take advantage of oil and gas opportunities in its waters, Greenland decided to invest in job training for skills that would also be useful in other industries, such as helicopter services and hotel services (AMAP, 2010). A similar approach might be useful in the fishing sector, for example training workers for a variety of maritime trades, although fishers will always be needed to catch the fish. Diversification beyond the fishing industry may also be a useful strategy.

4.6 Summary

Ocean acidification by itself appears to pose a modest threat to most Arctic and subarctic fisheries, especially in the next decade or two. Combined with ocean warming, however, acidification could have major effects in the longer term. Most of these are negative, such as declines in sea urchins in Norway and king crabs in Alaska and an increased likelihood of cod stock collapse in the Barents Sea, each of which undermines a fishery and thus livelihoods and food production. Some are positive, such as an overall increase in marine productivity in the Western Arctic Bioregion in Canada.

Management actions today can take into account the potential for disruption to marine ecosystems and populations and the fisheries that target them. Flexibility and adaptability are likely to be important characteristics of successful management systems. Enhancing the adaptive capacity of communities and local economies is also likely to be beneficial, for example encouraging diversification within and beyond the fishing sector. Bioeconomic models that can be used to explore plausible scenarios with confidence are unlikely to be developed for most fisheries and regions for some time to come. Monitoring of chemical and biological conditions in Arctic and subarctic waters, however, and research on increasing adaptive capacity in marine management and in economic planning can help in preparing for an uncertain future.

5. Conclusions, knowledge gaps and recommendations

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5.1 Overview

Since the publication of the first AMAP Arctic Ocean Acidification assessment in 2013, further research and data collection has reinforced the report's findings and deepened our understanding of the chemical, biological and socio-economic implications of ocean acidification in the Arctic.

The 2013 assessment reported that the Arctic Ocean and its marginal seas are acidifying – that is, their pH is falling – primarily due to the uptake of carbon dioxide (CO_2) from the atmosphere, but also as a result of other chemical additions and subtractions of carbon via other biogeochemical processes. The phenomenon, termed ocean acidification, is intensified in the Arctic relative to other regions of the world due to low ocean temperatures that increase gas solubility, and the inflow of naturally low pH water from the Pacific Ocean, river runoff and ice melt. The degree and distribution of acidification is not uniform across the Arctic Ocean, resulting in high temporal and spatial variability. This heterogeneity is driven by local and regional factors including sea-ice extent, freshwater inputs, ocean currents, water temperature and its effect on CO_2 solubility, the effects of biochemical processes and, hypothetically, by the extent of sub-sea methane seeps.

Chapter 2 of the 2018 assessment builds on the 2013 report by summarizing biogeochemical research published in the past five years. Acidification across the Arctic Ocean is increasingly evident, based on new and extended time-series. Focusing on seven Arctic regions, where there is sufficient information available, this chapter discusses trends and changes observed in the marine carbonate system and the regional drivers of ocean acidification. The chapter also discusses acidification in terms of the biologically-important measure of the saturation states of aragonite and calcite, two mineral forms of calcium carbonate widely used by Arctic organisms to build shells and skeletons, which declines in tandem with ocean pH.

The chapter also includes updated projections of Arctic Ocean acidification, noting that, with continuing carbon emissions, acidification is expected to continue progressing over the next century, although probably with considerable regional variability. The continued decrease in the saturation states of aragonite and calcite as a result of acidification, coupled with warming ocean conditions, is likely to have considerable impacts on marine ecosystems. Projections suggest that there will be significant changes in ocean chemistry in the coming decades, with the export of waters undersaturated in calcium carbonate from the Arctic increasingly impacting oceans outside the region (see Annex 1). Siberian shelves and the Canadian Arctic Archipelago already exhibit regional and seasonal aragonite undersaturation. Models project that surface waters in the Bering, Chukchi and Beaufort seas will be corrosive to aragonite between 2030 and 2050 under a business-as-usual emissions scenario. However, such conditions are not expected to be reached in the Barents and Norwegian seas until 2080.

It is anticipated that ocean acidification, particularly coupled with ocean warming and deoxygenation, will drive changes in marine ecosystems and impact Arctic biota. Anticipated biological changes, which are considered in Chapter 3, will be both direct and indirect. Likely direct effects include changes to key physiological aspects such as calcification, growth, reproduction, and metabolic rates, while indirect effects include changes in food availability and quality. Research suggests that direct and indirect effects will be highly heterogeneous with a mix of negative, positive and neutral responses. For example, there can be considerable response variability due to an organism's ability to adapt to ocean acidification or to changes in food availability. It is important also to consider that ocean acidification is taking place simultaneously with other environmental changes, such as warming, altered light availability, and the removal or introduction of key species.

Despite varied biological responses, complex ecosystem interactions among species and response uncertainty under multi-stressor conditions, it is likely that ocean acidification will drive changes at a magnitude that will affect people living in the Arctic and surrounding regions. These changes pose risks to commercial, subsistence, and recreational fisheries, as well as to the provision of other ecosystem services in the region.

Chapter 4 summarizes case studies commissioned by AMAP that explore the socio-economic impacts of ocean acidification in five Arctic regions. Three cases consider specific fisheries in the region, while two examine impacts and effects on regional commercial and subsistence fishing activities. Fisheries examined in the case studies include Norwegian kelp and sea urchins, Barents Sea cod and Greenland shrimp. Regional effects on commercial and subsistence fishing are examined in Alaska and in the western Canadian Arctic.

The case studies demonstrate that highly productive and locally important commercial and subsistence fisheries are vulnerable to the combined effects of acidification, warming, and other environmental stressors. While impacts on fisheries are likely to be modest in the next decade or two, they are likely to increase substantially beyond the middle of the century. Based on existing knowledge, not all fisheries impacts are considered major or necessarily negative; for example, warming and sea-ice retreat are likely to increase some fish stocks. However, it is unlikely that ecosystems and fish stocks will remain stable over the long-term.

The case studies illustrate that the effects of ocean acidification are not uniform across the Arctic and will therefore affect communities unevenly. Community resilience and adaptability is strongly tied to economic status and the ability of a community to create job diversity. As with biological impacts, the socio-economic effects are also taking place in the context of other social and economic changes, which are themselves leading to unpredictable impacts and interactions. The case studies demonstrate the high degree of uncertainty involved in anticipating changes caused by or relating to ocean acidification. Despite this uncertainty, the case studies show that successful fisheries management practices can be adapted to account for disruption due to acidification and other environmental stressors. Furthermore, economic planning and investment, such as training of workers, can help create greater economic diversity, adaptability and resilience in the face of uncertainty and change.

5.2 Knowledge gaps

Understanding of the processes driving ocean acidification continues to improve, although there remain important gaps in knowledge regarding chemical, biological, and socio-economic responses.

With respect to projected changes in ocean chemistry, much of this current understanding is based on global climate models rather than high-resolution regional models. Due to the high spatial heterogeneity of acidification across the Arctic and its surrounding seas, high resolution models will be of critical importance to improving understanding of region-specific processes and trends. Limited *in situ* observations of Arctic Ocean chemistry, particularly during winter months when the region is difficult to access, restrict our understanding of the system, especially in offshore regions.

Similarly, it is important to understand how individual biological species will respond and how species composition will change under multi-stressor conditions. This is particularly the case with organisms in the Arctic, which are less well studied than those elsewhere. Research should focus on biological responses of individual organisms, such as changes to metabolic performance, survival rates, growth rates, sex transition mechanisms, and reproduction, as well as on ecological and community-level changes, including predator conditions, food availability, and habitat. Of key importance is an understanding of the longer-term, multi-generational effects of modified environmental conditions on key species.

The cumulative new understanding of chemical and biological responses contributes considerably to our understanding of how economies and local societies are likely to be impacted by anticipated environmental changes and changes to the distribution, health and availability of important species. One uncertainty discussed in the 2018 report is how consumers in end markets will respond to changes in taste or texture of marine foods caused by ocean acidification.

Meanwhile, traditional and local knowledge has only been included in consideration of the effects of acidification to a very limited extent. Better use could be made of this knowledge, and indigenous people might be better engaged in regional, national and international processes, including coastal monitoring and research programs, in response to the effects of acidification and other anthropogenic climate change stressors on food security and well-being in the region.

5.3 Recommendations

There are specific actions that could be taken to improve the scientific understanding of processes related to ocean acidification. Enhancing research and monitoring of Arctic Ocean acidification must continue to be a high priority within the Arctic Council to promote cooperation between Arctic countries.

There is need for a unified monitoring program to harmonize and support adaptation actions in the Arctic and also to provide Arctic communities with the tools and training to conduct local, unified research and monitoring. Future decisions regarding use of living marine resources should take the many uncertainties still surrounding ocean acidification into account.

There is a need for more *in situ* research and observation to better understand the changing chemistry of Arctic waters, and Arctic-specific responses of biota. Based on the gaps identified here, future research should take a multi-stressor approach, given the inter-relationships, interactions and feedbacks between acidification and other stressors. Ecosystem changes should be monitored in such a way that allows identification and differentiation of the impact of each stressor on the ecosystem, as well as the potential synergistic effects of multiple stressors combined. This should also be extended to research conducted in the North Atlantic, given the biological, commercial and subsistence importance of fisheries in these waters and the impact of outflows from the Arctic basin. Laboratory research into physiological responses and genetic adaptation will be key to improving prediction of longer-term responses of biota to environmental change. Due to the scarcity of data and observations in the Arctic, a strong data sharing policy should be put in place and linked to global, open-source data depositories that can be accessed by modelers and the general public.

A lack of certainty about the interplay between biological changes and social and economic impacts of ocean acidification should not preclude action. Adaptation actions should be directed towards providing communities with flexibility, adaptability and economic and ecological resilience in the face of change and uncertainty: monitoring and investment decisions should aim to both reduce uncertainty and the costs of either underestimating or overestimating future impacts.

It is essential that action on adaptation is undertaken concurrently with mitigation. Ocean acidification mitigation is urgently required in order to avoid the most severe consequences that are projected in this report. Mitigation actions include both reducing anthropogenic carbon emissions and increasing carbon sink capacity.

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Annex 1

Export of calcium carbonate under-saturated waters from the Arctic Ocean to the North Atlantic

Authors: Leif G. Anderson, Kumiko Azetsu-Scott, Jon Olafsson

Contents

A1.1 Introduction

Ocean acidification has been a serious concern for the past two decades and is expressed by the decrease in pH in ocean waters that are in contact with an atmosphere of increasing carbon dioxide (CO_2). To date, this decrease is in the order of 0.1 pH units in the open ocean (Royal Society, 2005). However, ocean pH is variable due to many processes that influence the concentration of CO_2, including biochemical processes such as primary production and decay of organic matter, and physical processes such as changes in temperature with its impact on solubility of gases and circulation. Approximately half of the Arctic Ocean comprises shelf seas, which receive large volumes of river runoff. Biochemical processes are of considerable importance in coastal waters where the input from land adds terrestrial organic matter as well as nutrients, with the latter contributing to marine organic matter formation through primary production. The effect of biochemical processes on acidification state in the Arctic Ocean is also closely related to the general water mass transformation and circulation.

Waters from both the Pacific Ocean and Atlantic Ocean enter the Arctic Ocean and flow in a general circulation scheme before exiting to the Atlantic Ocean as illustrated in Figure A1.1.

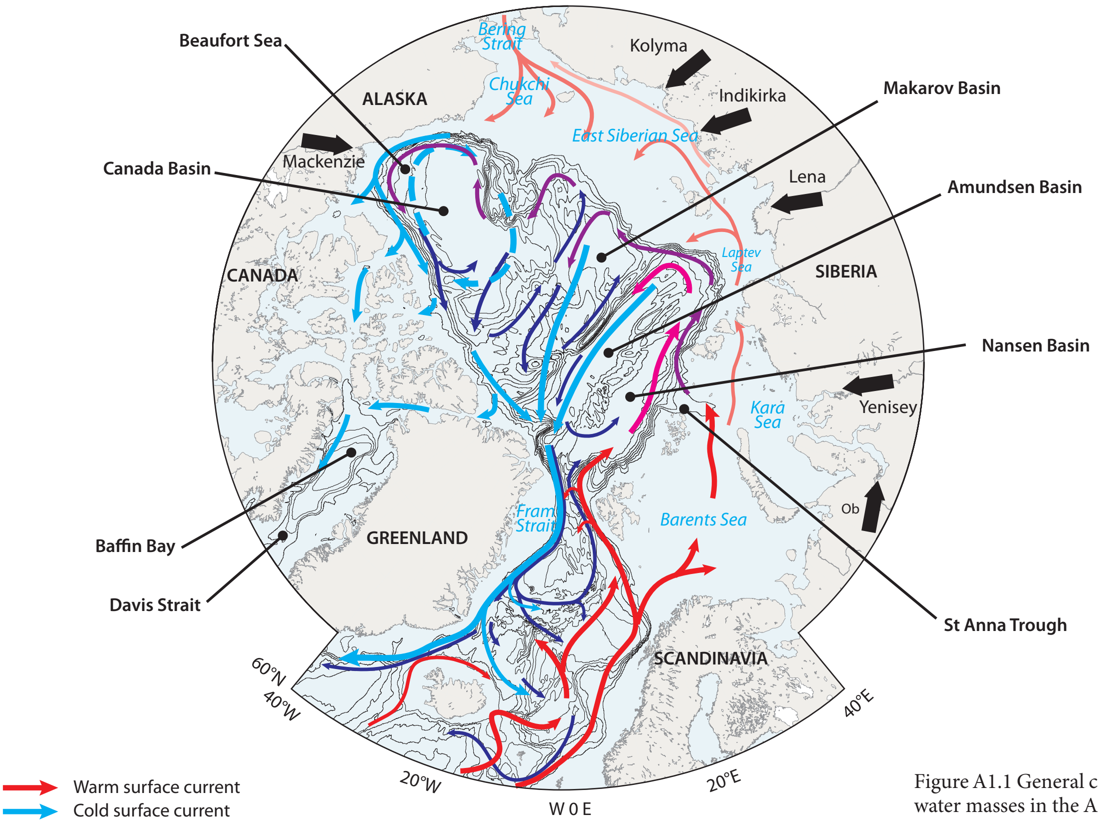


Figure A1.1 General circulation of water masses in the Arctic Ocean.

All of the Pacific water and a significant fraction of the Atlantic water passes over large shelves where substantial biochemical transformation occurs, especially during the summer months (Kaltin and Anderson, 2005).

The inflowing water from the Atlantic Ocean is saltier than that from the Pacific Ocean but has lower nutrient concentrations (e.g., Codispoti et al., 2013). The Atlantic inflow enters the Arctic Ocean via two branches, the Fram Strait branch that flows north along the west coast of Svalbard and the Barents Sea branch that flows through the Barents Sea into the Kara Sea where it enters the central Arctic Ocean through the St Anna Trough (e.g., Schauer et al., 2002). Part of the Atlantic water continues through the Vilkitsky Strait into the Laptev Sea, while mixing with river runoff. Even if the nutrient concentration is not as high in the Barents Sea as it is in the Chukchi Sea, new primary production is still high at up to 100 gC/m^2/y in some areas (Sakshaug, 2004).

Subsurface waters flowing into the Arctic Ocean through Bering Strait have high nutrient and CO_2 concentrations because they have been exposed to mineralization of organic matter. Thus large quantities of nutrients enter through Bering Strait and supply both the Chukchi Sea and East Siberian Sea. Historically the Chukchi Sea has been largely ice-free during summer, but in recent years this trend has also extended to the East Siberian Sea. Hence primary production is substantial and the Chukchi Sea is reported to be the Arctic shelf sea with the highest new primary production, up to 160 gC/m^2/y (Sakshaug, 2004). Levels reported for the East Siberian Sea are significantly lower but lack of measurements makes this estimate highly uncertain.

Two dominating regimes exist in the East Siberian Sea: a western part dominated by waters from the Laptev Sea, mainly of Atlantic Ocean and river water origin, and an eastern part with a clear Pacific Ocean influence (e.g., Pipko et al., 2011). It is likely that productivity is higher in the eastern domain because the supply of nutrients is much higher from the Pacific Ocean than from the Atlantic Ocean, especially after the water has passed long distances over the shelves. The location of the interface between the two domains is determined by the winds that are set by the dominating atmospheric pressure field.

In the shallow Siberian shelf seas, mostly less than 50 m deep, the majority of the plankton ends up at the sediment surface when they die in the autumn. Added to this, organic matter is arriving from land via both river runoff and coastal erosion (e.g., Alling et al., 2012; Karlsson et al., 2016), a fraction that is likely to increase as permafrost thaw accelerates under climate change. However, the terrestrial input is much less in the most eastern part of the East Siberian Sea as seen from the sediment record of $\delta^{13}C_{org}$ (e.g., Anderson et al., 2009), the area highly impacted by Pacific water.

Regardless of the organic matter source, bacterial degradation produces CO_2 and nutrients that leak out from the sediment into the bottom water during the winter months. Sea ice also forms at this time, a process whereby salt is expelled as brine and enters the underlying water. In these shallow seas the brines contribute to a relatively high-salinity bottom water, at least in regions where the surface salinity is not too low as a result of river water contribution. Consequently, the organic matter decay products are concentrated in this high-salinity shelf water that has been observed to flow off the shelf at a salinity of around 33 in both the East Siberian Sea and Chukchi Sea (e.g., Bates, 2006; Anderson et al., 2010; 2017; Torres-Valdés et al., 2013).

The most prominent signature of this water, rich in decay products, has historically been silicate. This was first reported from the ice island T-3 in 1968/1969 (Kinney et al., 1970) as a maximum at about 150 m depth in Canada Basin. The silicate maximum was associated with phosphate and nitrate maxima as well as oxygen and temperature minima. The temperature was close to freezing indicating a winter source. In 1979 the same high silicate feature was reported over the Lomonosov Ridge all to the flank into the Amundsen Basin during the LOREX ice camp (Moore et al., 1983). Historically the high nutrient signature, predominantly silicate, found at a salinity of about 33 has been observed in the out-flow regions to the Atlantic Ocean, such as at the tip of the Morris Jesup Plateau (Anderson et al., 1994) in the western Fram Strait (e.g., Anderson and Dyrssen, 1981) and in the Canadian Arctic Archipelago (e.g., Codispoti and Owens, 1975; Jones and Coote, 1980).

Elevated dissolved inorganic carbon concentrations were first reported in this high nutrient water during the CESAR ice camp 1983 (Jones and Anderson, 1986). Since then many studies have observed this high nutrient water and its accompanying high fCO_2, low pH and low calcium carbonate saturation state in the Pacific sector of the Arctic Ocean (e.g., Jutterström and Anderson, 2005; Mathis et al., 2007; Bates et al., 2009; Yamamoto-Kawai et al., 2009; Anderson et al., 2010).

This case study examines the ocean acidification conditions, expressed as calcium carbonate saturation state, in the waters that enter the Arctic Ocean, how these conditions are affected by processes within the Arctic Ocean and how these signatures are distributed in the deep basins, as well as how the water of low calcium carbonate saturation state exits into the Atlantic Ocean. The concentration of hydrogen ions, normally stated as pH ($-\log[H^+]$) is determined by several chemical equilibria, among them, that of carbonic acid normally has the greatest impact in the marine environment. The calcium carbonate saturation state (Ω) is determined by the concentrations of carbonate ion (set by the pH and the dissolved inorganic carbon concentration, for example) and calcium ion, where the latter is mainly a function of salinity. Consequently, calcium carbonate saturation state is not only dependent on pH but gives a good indication of the ocean acidification condition. This annex uses the calcium carbonate saturation state as it gives a clear representation of conditions, with values above 1 depicting supersaturation and values below 1 depicting undersaturation. Here, calcium carbonate saturation state is exemplified by the saturation state of aragonite (Ω_{arg}) – one of the two most common naturally occurring forms of calcium carbonate.

A1.2 Methodology and data

This contribution is based on data from several cruises (Table A1.1) performed in the Arctic Ocean and its gateways. Data used include temperature and salinity (from CTD

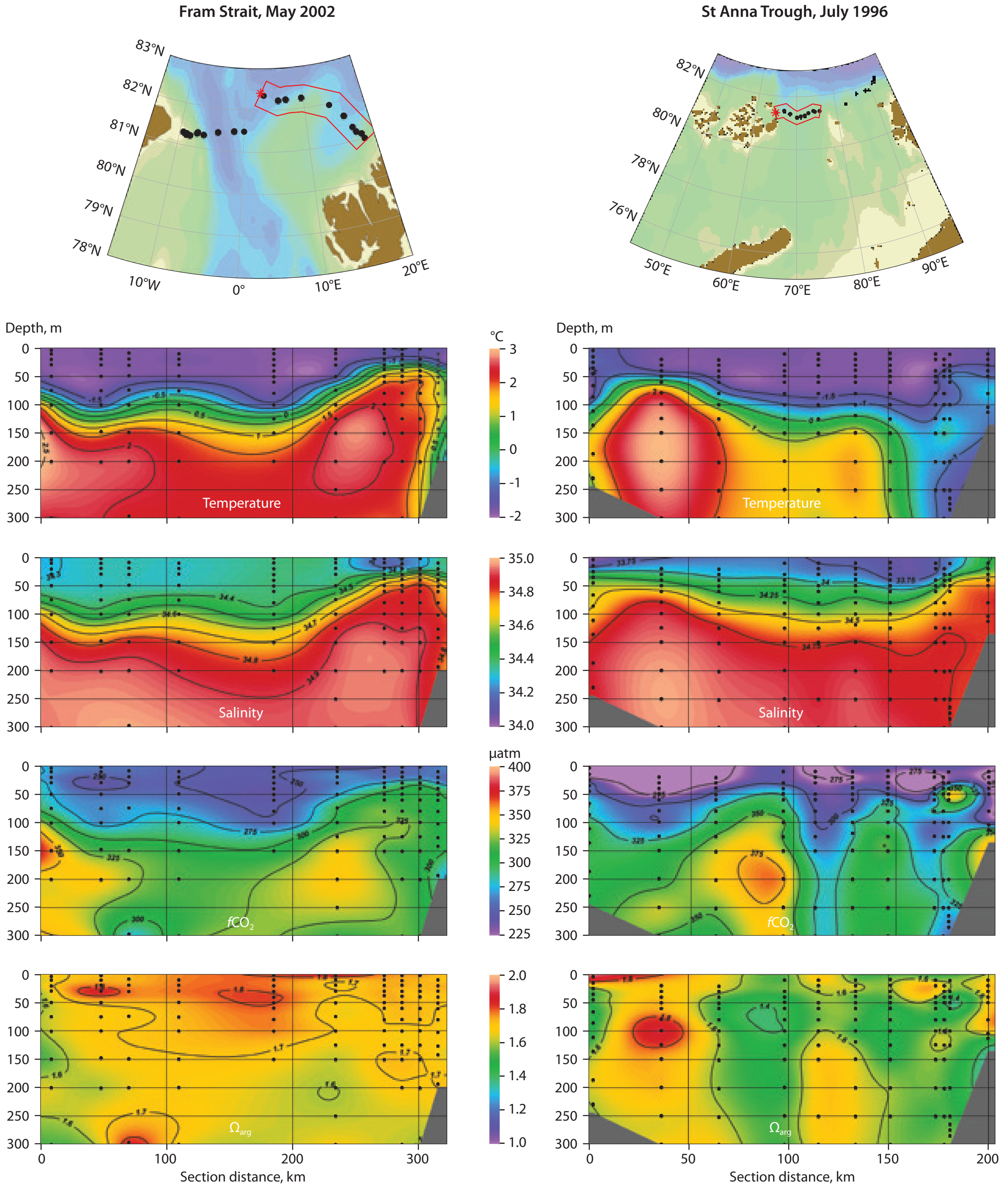


Figure A1.2 Properties in the upper 300 m of the inflowing Atlantic water as observed in Fram Strait in May 2002 and the St Anna Trough in July 1996, for the stations noted on the locator maps.

profilers), silicate, and carbon-system parameters. The fugacity of carbon dioxide ($f\mathrm{CO_2}$) (where the fugacity corresponds to partial pressure of CO_2 but also takes account of CO_2 not being an ideal gas) and saturation state of aragonite (Ω_{arg}) are computed from two parameters of the carbonate system; if pH was determined it was used together with total alkalinity, and if not then dissolved inorganic carbon and total alkalinity were used (see Table A1.1). On most cruises the analytical methods used allow for accuracy in $f\mathrm{CO_2}$ and Ω_{arg} of ±10 µatm and 0.05, respectively.

A1.3 Results and discussion

Inflowing water from the Atlantic Ocean has a salinity close to 35 and a temperature higher than 2°C, the latter varying with season. Once this water meets the sea ice, water temperature lowers to near freezing and salinity is lowered to around 34 in the upper 50–100 m due to ice melt. This can be seen in both the West Spitsbergen Current north of Svalbard and in the St Anna Trough (Figure A1.2) where the Barents Sea Branch enters the central Arctic Ocean (e.g., Schauer et al.,

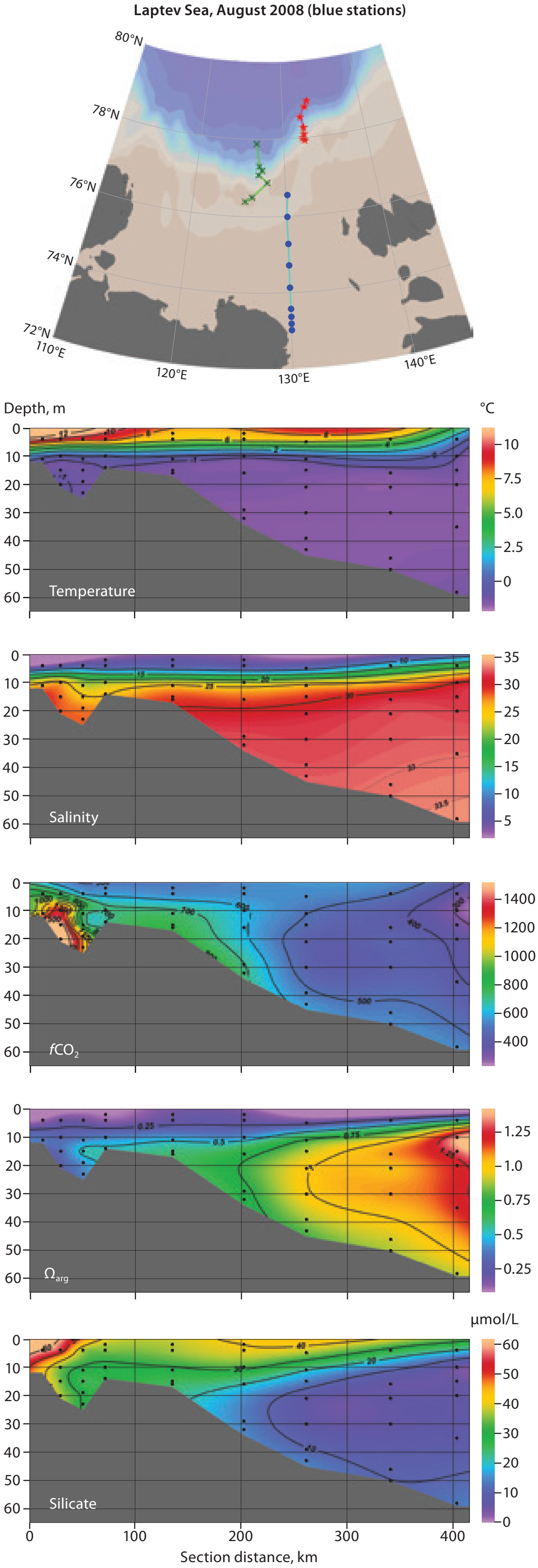
Table A1.1 Cruises from which data are used.

Cruise	Year	Ship	Carbon-system parameters
Oden 91	1991	*Oden*	TA, DIC
ACSYS	1996	*Polarstern*	TA, DIC, pH
OA-02	2002	*Oden*	TA, DIC, pH
Smith Sound	2003	*USCGC Healy*	TA, DIC
Davis Strait	2005	*CCGS Hudson*	TA, DIC
Barrow Strait	2005	*NGCC Des Groseilliers*	TA, DIC
Beringia	2005	*Oden*	TA, DIC, pH
ISSS-08	2008	*Yacob Smirnitskyi*	TA, DIC, pH
SWERUS-C3	2014	*Oden*	TA, DIC, pH

TA: total alkalinity; DIC: dissolved inorganic carbon.

2002). Ω_{arg} is around 1.7 in both the cold surface water and the warmer underlying water (Figure A1.2). Observed fCO$_2$ in the warm Atlantic water (temperature around 4°C) west of Svalbard was seen to be in the range 325–350 µatm during January–February and November 1999 (Nakaoka et al., 2006). In 1996 and 2002, the surface water at a temperature close to freezing had an fCO$_2$ of about 275 µatm (Figure A1.2) which is close to what is expected from the cooling as CO$_2$ solubility increases with decreasing temperature. The atmospheric increase in fCO$_2$ of just under 10 µatm during the period of close to 10 years of these observations is not visible in the variability of the data. These observations are in reasonable agreement with the fCO$_2$ data from the N-ICE expedition north of Svalbard in 2015, where researchers observed values at 5 m under the ice from around 270 µatm in January, slowly increasing up to ~300 µatm in May followed by a sharp decrease when the spring bloom set in (Fransson et al., 2017). Observations from the Svalbard area to the Laptev Sea in late August to early October in the years 2006, 2007 and 2009 showed considerable patchiness in fCO$_2$ values, however these values were always undersaturated except for regions highly impacted by river runoff (Pipko et al., 2017). Hence the waters flowing in from the Atlantic Ocean are well supersaturated with respect to aragonite.

The upper water flow that enters the central Arctic Ocean from the Atlantic Ocean, both through Fram Strait and through the Barents Sea continues east along the continental margin towards the Laptev Sea. In summer much freshwater is added by sea ice melt and from the main Siberian rivers (including the Ob, Yenisei and Lena) making the surface layer less saline. The flow pattern of the runoff on the shelf largely depends on the wind field and as this varies over time so too does the flow pattern. Hence, the wind field is also a factor determining where the freshwater plume enters the deep central Arctic Ocean. The runoff significantly impacts the biogeochemistry of the system, which is clearly seen within the Laptev Sea where with decreasing salinity both silicate and CO$_2$ increase in the surface water (Figure A1.3). The silicate concentration is relatively high in Siberian rivers (e.g., Cauwet and Sidorov, 1996) from erosion of minerals in the drainage basins. These rivers are also loaded with organic matter that is mineralized, resulting in high fCO$_2$. This together with decreasing calcium concentration in low salinity waters also lowers the Ω_{arg}.



Laptev Sea, August 2008 (blue stations)

Figure A1.3 Properties along a section from the Lena delta to the outer Laptev Sea in 2008.

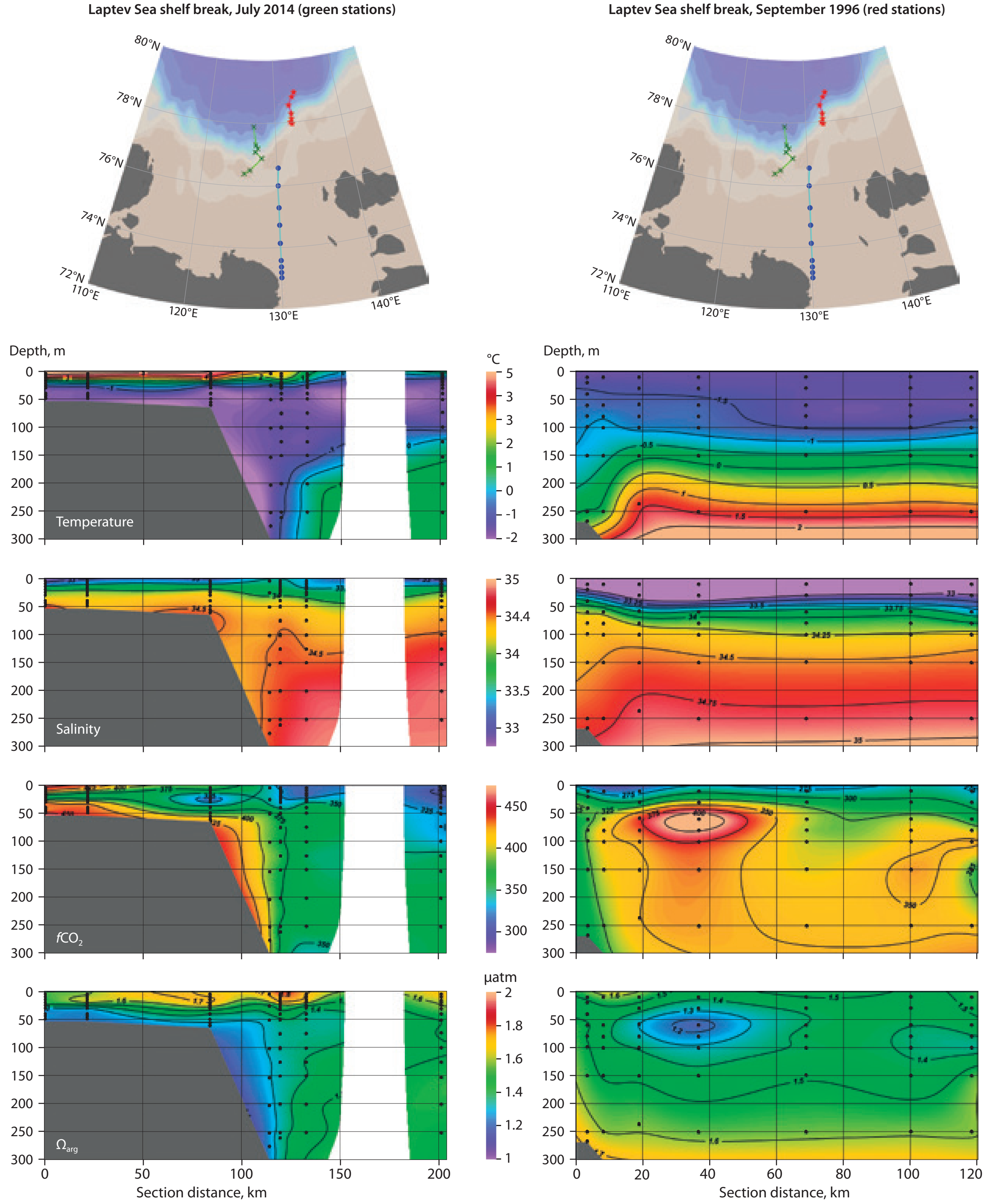


Figure A1.4 Properties in the upper 300 m at the Laptev Sea shelf break in 1996 and 2014, for the stations noted on the locator maps.

At the Laptev Sea shelf break the fCO_2 and Ω_{arg} levels are close to the same as in Fram Strait and the St Anna Trough, even if the Ω_{arg} has decreased somewhat and also shows signs of subsurface minima (Figure A1.4). Data from 1996 and 2014 show some similar patterns, but also large differences that are probably due to the differences in sea ice conditions, with the ice cover being extensive in 1996 but with open water in 2014. This is evident in the surface water temperature which was close to freezing in 1996 but up to ~5°C in 2014 (Figure A1.4). In 1996, a thicker low salinity surface layer was also present. In 2014, the inner stations have a CO_2 supersaturated surface layer, which is likely to be due to mineralization of organic matter in the river runoff as also seen closer to the Lena Estuary in 2008 (Figure A1.3). Furthermore, the bottom water was supersaturated and had a Ω_{arg} minima indicating that remineralization at the sediment surface is the responsible process, and that this water can extend out over the deep basin.

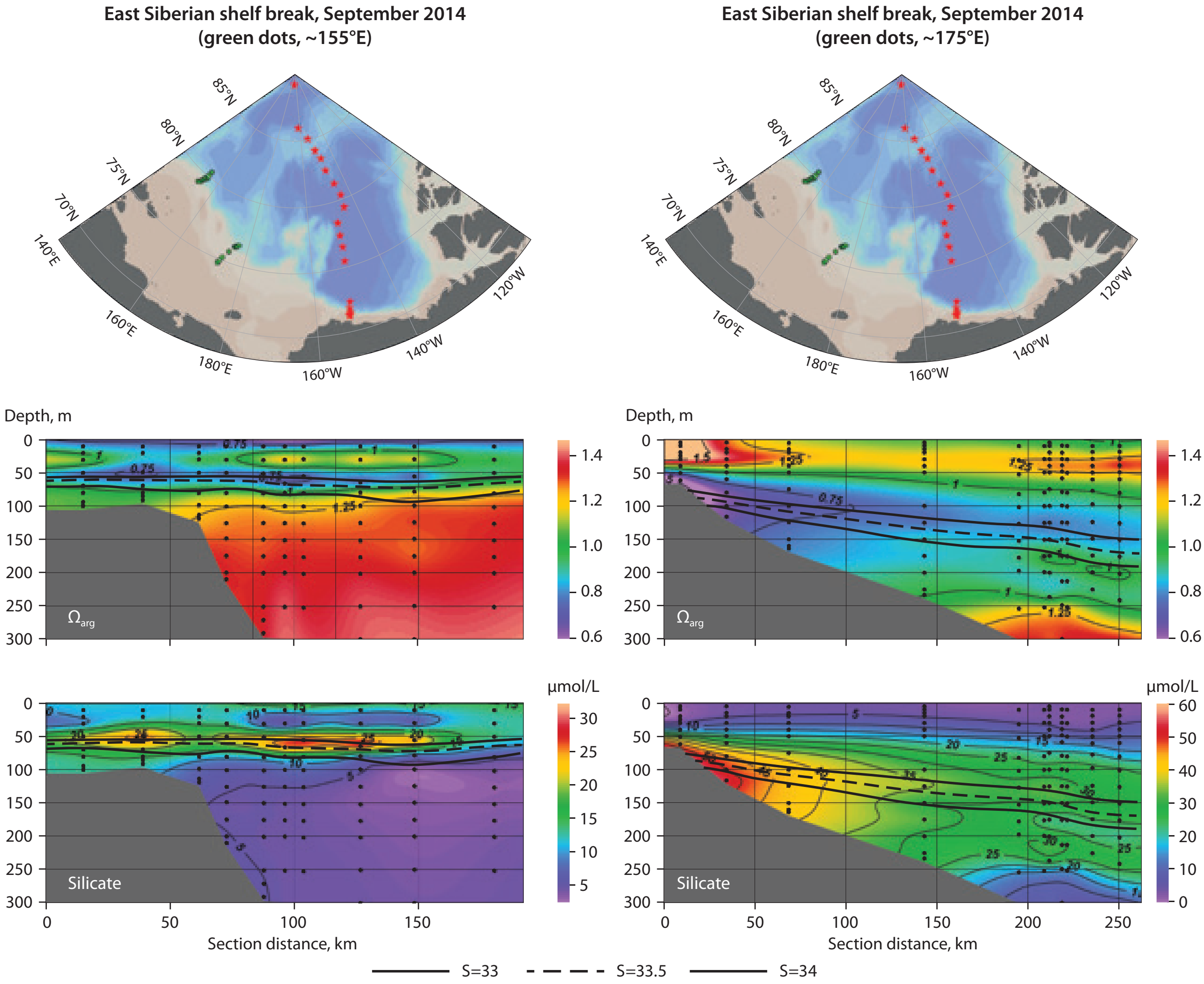


Figure A1.5 Properties across the East Siberian shelf break, September 2014, for the stations noted on the locator maps.

Moving further into the East Siberian Sea the impact of mineralization in the shelf bottom water is even more pronounced (Figure A1.5). The East Siberian Sea and the Chukchi Sea are the formation areas of the nutrient maximum found in the central Arctic Ocean at a salinity of around 33, see Section A1.1. It is also evident that the maximum in silicate is strongly associated with a minimum in Ω_{arg}, both at the shelf break and within the central Canada Basin (Figure A1.6). The *f*CO_2 is over 1000 µatm in some locations, namely at levels projected for the atmosphere in 2100, and thus has a corresponding saturation state.

The high *f*CO_2 halocline water is overlaid by a fresh surface layer, built up by runoff and ice melt. This surface layer prevents the supersaturated CO_2 in the halocline from equilibrating with the atmosphere and this water, strongly undersaturated in aragonite, can thus be transported far from this formation region and still keep much of its signature. Only mixing will increase Ω_{arg}. This is opposite to the supersaturated *f*CO_2 in the surface water that comes from river runoff (Figure A1.3). This water maintains some of its signature as long as the sea ice cover persists, but with more open water it will soon equilibrate with the atmosphere and achieve the classical anthropogenic ocean acidification, namely the level that corresponds to equilibrium with an atmosphere of anthropogenic CO_2.

High quality data on the carbon system started to be collected in the 1990s and the extent, both horizontally and vertically of the aragonite undersaturated water has since increased (Qi et al., 2017). Much of this increase is caused by the addition of more 'Pacific Winter Water', namely bottom water from the shelf with a high content of organic matter mineralization products. The extent of the aragonite undersaturated region is also influenced by the increased anthropogenic atmospheric CO_2 concentration. The impact of anthropogenic CO_2 was evaluated by Anderson et al. (2010) who estimated the anthropogenic carbon concentrations from SF_6 and CFC-12 data using the transient time distribution (TTD) method (Tanhua et al., 2009). The result was a substantial increase in the extent of the aragonite undersaturated ($\Omega_{arg}<1$) region from pre-industrial times to 2005. Assuming a continued exponential increase in anthropogenic carbon in the atmosphere and an unchanged Arctic Ocean circulation pattern, the aragonite undersaturated water will further increase to 2050 at which point it may even reach the surface (Anderson et al., 2010).

As seen from Figures A1.5 and A1.6 the water undersaturated in aragonite and high in silicate is largely confined to Canada Basin. However, it is well-known that the horizontal extent of the silicate maximum in the upper halocline has varied

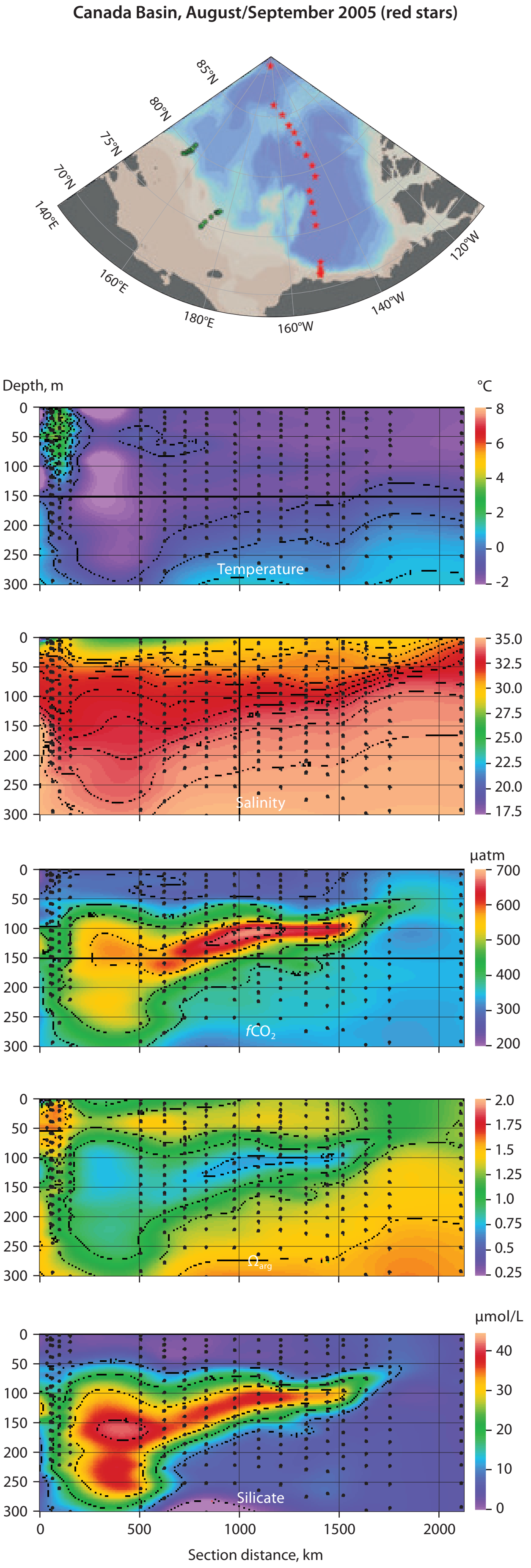


Figure A1.6 Properties across Canada Basin in August/September 2005, for the stations noted on the locator map.

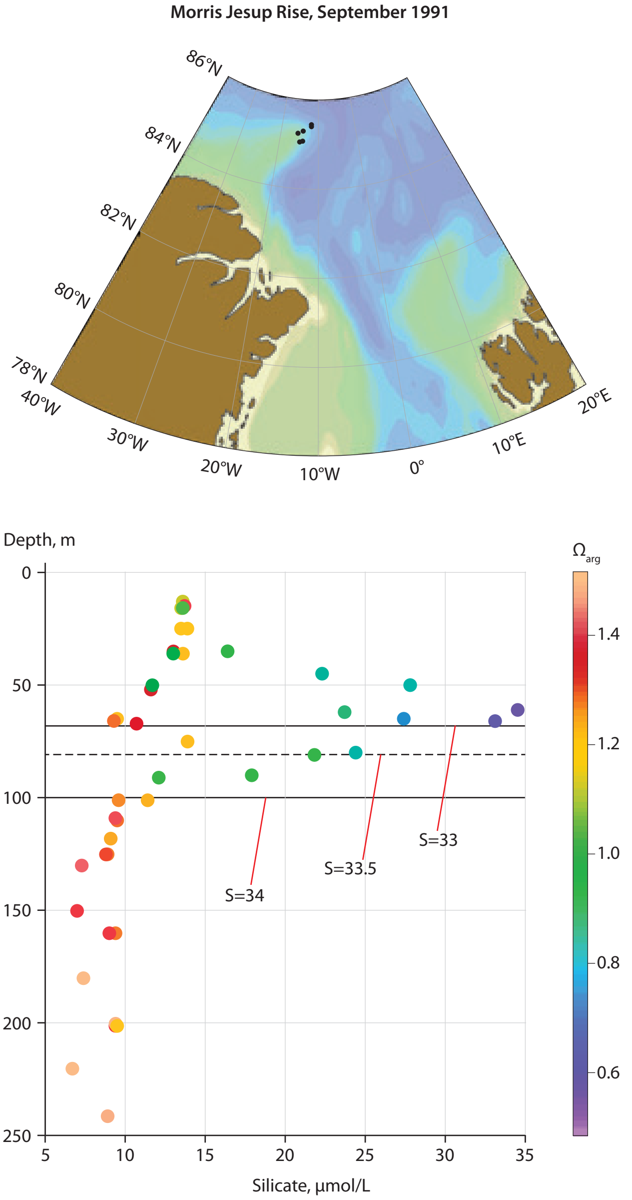


Figure A1.7 Silicate profiles, color coded by Ω_{arg}, at the tip of the Morris Jesup Rise in 1991.

substantially over time and was for instance observed over the Lomonosov Ridge in 1979 (Moore et al., 1983). It is likely that this water also had low Ω_{arg} but to what degree is not known. The areal coverage of the silicate maximum is likely to be coupled to the extent of the Beaufort Gyre, which in turn is dependent on the dominating atmospheric pressure field, often expressed by the Arctic Oscillation (e.g., Proshutinsky et al., 2015). It has been suggested that during negative Arctic Oscillation indices most of the nutrient maximum water exits to the Atlantic Ocean through the Canadian Arctic Archipelago, while during positive indices a substantial fraction exits through Fram Strait (Steele et al., 2004). Hence, the atmospheric pressure field is crucial in determining the fate of the large volume of halocline water undersaturated in aragonite.

Halocline water of high silicate concentration and aragonite undersaturation (Ω_{arg}<1) was observed at the tip of the Morris Jesup Rise in 1991 (Figure A1.7). This is a region where the

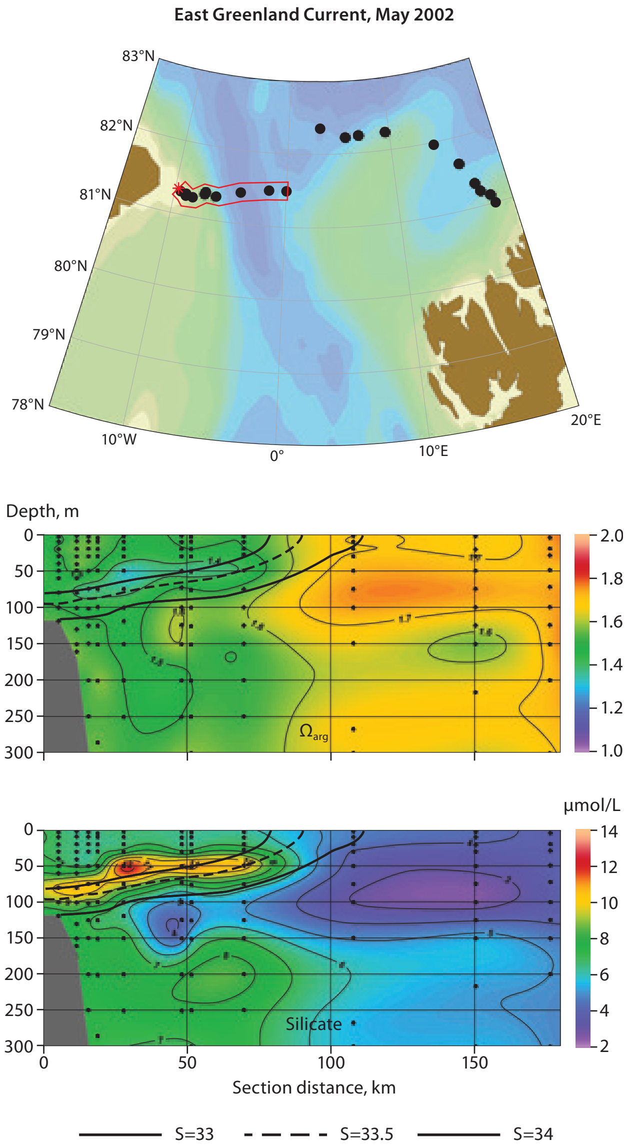


Figure A1.8 Properties across the East Greenland Current in May 2002, for the stations noted on the locator map.

outflow of water from the Beaufort Gyre passes on its way to the North Atlantic east of Greenland (Rudels et al., 1994). Waters of high silicate concentration and a salinity of around 33 were observed on several cruises in the East Greenland Current, such as in 1980 (Anderson and Dyrssen, 1981), 1984, 1990 and 1997, but not 2004 (Falck et al., 2005) or 2005 (Torres-Valdés et al., 2013). It has been suggested that the extent of the Beaufort Gyre might shift under different atmospheric circulation regimes expressed by the Arctic Oscillation index (e.g., Steele et al., 2004; Proshutinsky et al., 2015), which could in turn imply that the outflow of this water through Fram Strait relates to the Arctic Oscillation. Even if no conclusive evidence for this has been reported, the interannual observations of the silicate concentration in the East Greenland Current could be related to long-term shifts in the Arctic Oscillation.

Unfortunately, few publications on the biogeochemistry in Fram Strait report the saturation state of calcium carbonate.

In May 2002, corresponding to the end of the period with observed high silicate concentrations, hydrographic observations were carried out along the East Greenland Current from ~81.5°N to 65°N south of Denmark Strait (Jones et al., 2008). The carbonate system was determined and a weak maximum in silicate was observed in the East Greenland Current which coincided with a minimum in Ω_{arg} (Figure A1.8). It is useful to compare this minimum of just below 1.3 in Ω_{arg} with observations in the inflowing water of the eastern Fram Strait. Even if no undersaturation was observed in the water flowing out of the Arctic Ocean at this time it was significantly less saturated, up to 0.5, than the inflowing Atlantic water (Figure A1.2). Hence processes in the Arctic Ocean have the potential to influence conditions in the outflow regions of the Atlantic Ocean to a degree that amplifies the anthropogenic signal. In the outflow region the Ω_{arg} minimum is over the continental slope, the region where the water of Pacific origin flows out from the Arctic Ocean. This is the core of the East Greenland Current and can be traced all the way south to Denmark Strait (Jones et al., 2008; Sutherland et al., 2009). However, the nutrient maximum as well as the Ω_{arg} minimum signature fades away along the flow path by mixing as well as by seasonal biochemical processes and air-sea interaction.

The Canadian Arctic Archipelago and Hudson Bay are major pathways of water flow from the Arctic Ocean to the Northwest Atlantic. Based on measurements of dissolved inorganic carbon and total alkalinity in samples collected over the period 2003–2005 in different parts of the Canadian Arctic Archipelago, low pH and calcium carbonate saturation states were reported (Azetsu-Scott et al., 2010). The water at salinity ~33 had aragonite saturation minima both in Barrow Strait and Smith Sound (Figure A1.9). These minima were associated with high silicate concentrations (Figure A1.9). In Davis Strait low Ω_{arg} occurs at the western side of the section, extending towards East Greenland Slope, corresponding to high silicate concentrations. In all three sections, low aragonite saturation states (Ω_{arg}<1.2) are found in the Arctic outflow (Azetsu-Scott et al., 2010). The Arctic outflow through the Canadian Arctic Archipelago has a high proportion of Pacific waters (Azetsu-Scott et al., 2012), which have a low saturation state. The Pacific waters are further modified by processes such as mixing, biological activity, ice formation and melt, and river runoff in the Arctic before they flow out and to Baffin Bay. The Arctic outflow waters with low Ω_{arg} can be traced along the western Baffin Bay to Davis Strait. The water from the Hudson Bay with low Ω_{arg} due to the large river input (Azetsu-Scott et al., 2014), mixes with the outflow through Canadian Arctic Archipelago and together they flow further south to the Northwest Atlantic.

It should also be noted that the waters around Greenland are exposed to freshwater input from the ice sheet, which will lower the saturation state of the water with respect to calcium carbonate. With increasing glacial melting this impact will become greater and could potentially be added to the effect in the water flowing out of the Arctic Ocean, especially on the east coast where the outflow follows the Greenland coast.

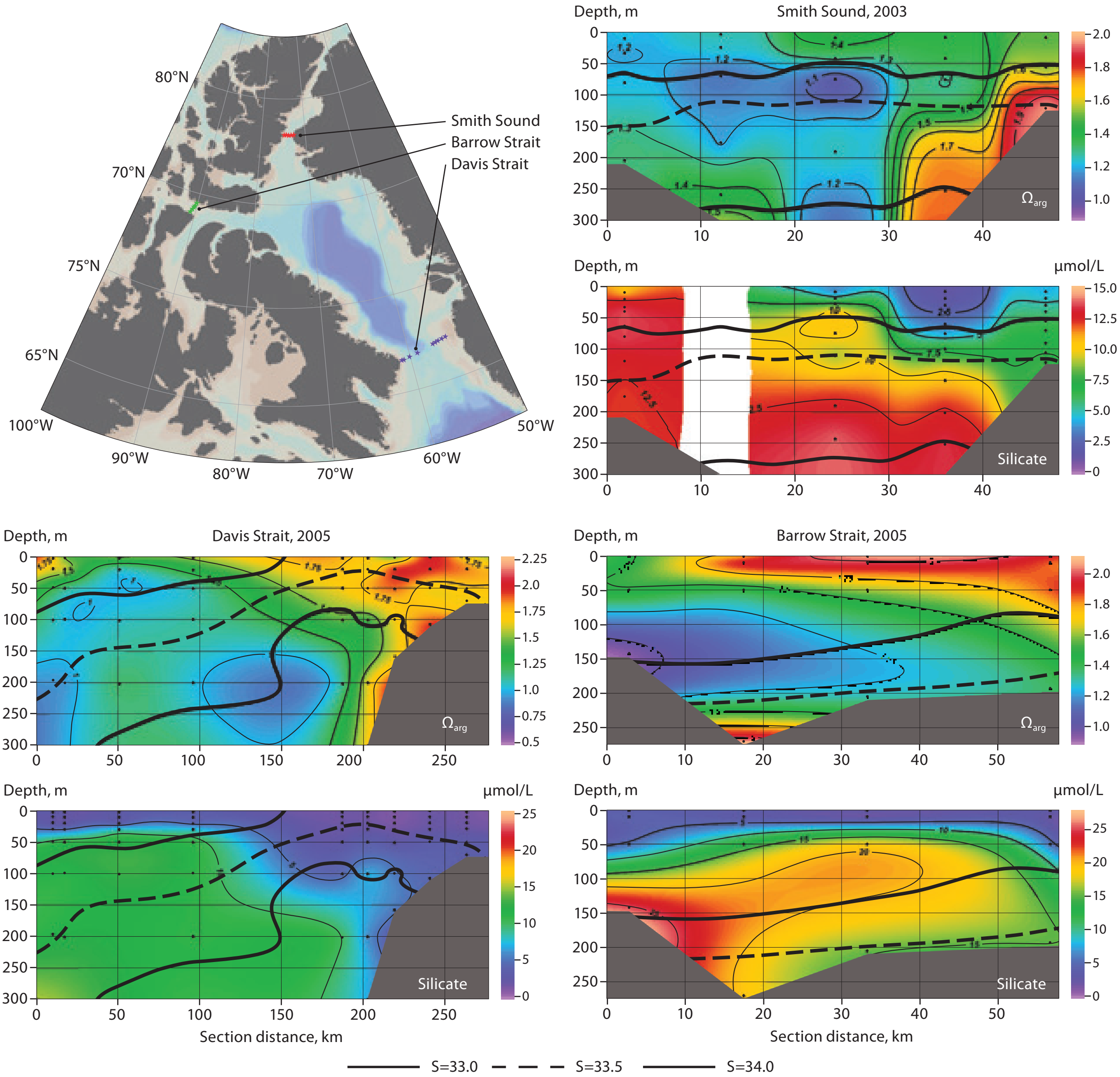


Figure A1.9 Properties across Barrow Strait in 2005, Smith Sound in 2003 and Davis Strait in 2005, for the stations noted on the locator map.

A1.4 Conclusions and summary

The conditions relevant to the acidification in the upper waters of the Arctic Ocean and the export of those waters to the Atlantic Ocean are summarized in Figure A1.10, and can be described as follows. Waters with high fCO_2 are formed in the Chukchi Sea and East Siberians Sea, where the chemical signature is formed through degradation of organic matter, of both marine and terrestrial origin, mainly at the sediment surface (e.g., Semiletov et al., 2016). Much of this water originates from the Pacific Ocean and is already rich in fCO_2 from processes in the North Pacific when entering through Bering Strait. On the shelves sea ice is formed every winter, a process that expels brine which sinks to the bottom of these shallow seas forming a high salinity bottom water to which the CO_2 and other products from the organic matter degradation are added. The shelf water, with fCO_2 substantially higher than the atmospheric value and with the core being even higher than what is projected for the atmosphere in 2100, exits the shelf into the deep Canada Basin. Here the water forms the upper halocline and may be 'stored' for several decades in the Beaufort Gyre at a depth of around 100 m, from which there is an outflow of water that is exported to the North Atlantic both to the west and east of Greenland. The inflow from the Atlantic on the other hand is undersaturated with respect to atmospheric CO_2, a condition that is caused by a combination of primary production and cooling that both decrease fCO_2.

The high fCO_2 shelf water is also undersaturated with respect to calcium carbonate, an effect that is enhanced by input of freshwater from river runoff and sea ice melt. This condition leads to calcium carbonate undersaturated surface waters in regions of high sea ice melt as well as outside river mouths. In addition, much surface water of low salinity is 'trapped' in the Beaufort Gyre. With rapid warming in the Arctic, more sea ice melts and the discharge from land increases as precipitation intensifies, leading to larger areas of fresh surface waters of

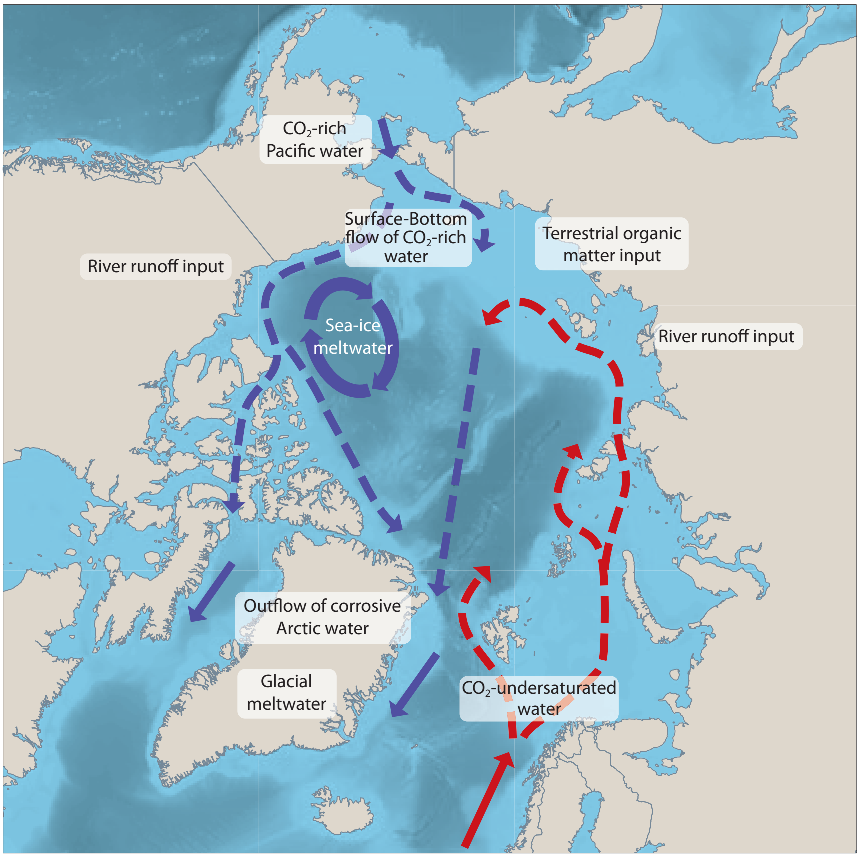


Figure A1.10 Summary of conditions relevant to acidification in the upper waters of the Arctic Ocean and its export to the Atlantic. The arrows indicate the general flow of the upper waters.

low calcium carbonate saturation state. Accelerated thawing of permafrost leads to more supply of organic matter to the shelf seas that result in even higher fCO_2 through mineralization of this organic matter. Hence the fCO_2 levels of the upper halocline might increase further in the future with even more extensive waters that are undersaturated with calcium carbonate.

Since the shelf regions of the North Atlantic flooded by the Arctic Ocean outflows are both biologically productive and support important commercial fisheries, continued monitoring of the changes in ocean acidification state and investigations of biological responses to ocean acidification in this area are urgently needed.

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Annex 2

Urchin harvesting and kelp regrowth in northern Norway under ocean acidification and warming

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Contents

A2.1 Introduction

Rising carbon dioxide (CO_2) concentrations in the atmosphere are driving important changes in the oceans. More CO_2 is being absorbed by the ocean, causing chemical changes referred to collectively as 'ocean acidification' (Feely et al., 2004; IPCC 2014). Through the greenhouse effect, increasing atmospheric CO_2 is also causing a rise in global air temperature (often termed 'global warming') which in turn leads to warmer oceans. Such changes can affect marine biota and ecosystems, potentially modifying the provision of ecosystem services to human societies such that human welfare is impacted (Doney et al., 2009; Brander et al., 2014; Falkenberg and Tubb, 2017).

Coastal areas at high latitudes such as northern Norway (>65°N) may be particularly vulnerable to ocean acidification. Carbonate concentrations and seawater buffering capacity are naturally lower here due to low temperatures and riverine/freshwater inputs. Coastal acidification can also be locally exacerbated by eutrophication (Cai et al., 2011; Wallace et al., 2014), coastal upwelling (Feely et al., 2008; Haigh et al., 2015), and the 'biological amplification' effect of remineralization in shallow bottom waters (Bates and Mathis, 2009). Northern Norway and the Barents Sea have also been identified as hotspots of ocean warming in observations over the past three decades (Levitus et al., 2009) and century-scale model projections (Biastoch et al., 2011; Bopp et al., 2013; Renaud et al., 2015).

Under contemporary conditions, coastal areas provide a range of ecosystem services to human societies including provisioning, in particular of animals harvested for food (Ferreira et al., 2016). Within northern Norway an emerging fishery is that for the green sea urchin (*Strongylocentrotus droebachiensis*). High demand on the global market combined with a global shortage of supply and sequential expansion of the harvested area, offer the opportunity to develop a potentially profitable industry (Berkes et al., 2006; Chen and Christie, 2016). In some areas, however, urchins may be regarded as a pest that removes valuable kelp forest (notably sugar kelp *Saccharina latissimi*) through overgrazing, and there may be an interest in culling the urchins to allow kelp regrowth. This regrowth can be beneficial as kelp also provides a range of ecosystem services. For example, in Norway kelp has been commercially harvested for more than 50 years (Vea and Ask, 2011). Kelp forests can also provide nursery grounds, feeding areas or migration routes for fish species of economic value (e.g., coastal cod *Gadus morhua*; Norderhaug and Christie, 2009; Seitz et al., 2013) as well as other ecosystem services such as carbon sequestration, water quality regulation, and cultural services such as diving and recreational fishing (Falkenberg and Tubb, 2017).

Ocean acidification and warming are emerging as external factors that can modify fisheries and their management (Anderson et al., 2011). It is anticipated that, in the future, managers will need to act to avoid the potentially pervasive negative effects of climate change on a range of economic properties of fisheries, including net present value, revenue, economic yield, and total profits (reviewed by Falkenberg and Tubb, 2017). It may be necessary to modify human harvesting activities such that management of natural resources is optimized.

This study explores optimal harvesting strategies for the emerging urchin fishery in northern Norway, and urchin culling requirements to allow the regrowth of kelp forest. The specific questions asked are: What are the potential impacts of ocean acidification and warming on urchin harvest yield and what are the optimal management strategies for urchin harvesting and culling? A simple kelp-urchin dynamical model is used to explore management strategies in which exhaustive harvesting (or culling) is allowed within a specified area but the minimum harvested (or culled) urchin size is restricted. These simulations allow size limits to be identified that maximize harvest yield or ensure an effective cull during present-day (2000–2020) and future periods (2030–2050 under the SRES A1B scenario).

A2.2 Methods

The dynamics of sugar kelp and green sea urchins were modeled in a non-retentive fjord in northern Norway. The approach loosely follows that of Marzloff et al. (2013) (see Appendix Table A2.1 for details). Briefly, the kelp is modeled as a bulk biomass with logistic growth dynamics, a constant recruitment flux from settling sporelings, and grazing losses proportional to urchin biomass (Eqns A1, A3; Larson et al., 1980; Meidel and Scheibling, 1999). The urchin population is age-structured in classes 0–1, 1–2, …, 14–15 years, neglecting the biomass contributions of individuals >15 years (Fagerli et al., 2015). To account for size-dependent grazing/predation/harvesting effects, the mean and standard deviation of test diameter within each age class are modelled dynamically (Eqn A7) and used to simulate individual test diameters and body/gonad masses (Eqns A8–A10). Urchin recruitment to the 0–1 age class varies stochastically between years and is strongly limited by the presence of kelp (Eqns A2, A4; Fagerli et al., 2013, 2015). Survival between age classes follows a constant, size-independent mortality rate and a predation mortality rate that increases with kelp cover and is reduced for larger individuals (large size refuge) (Eqns A2, A5, A6). Net immigration of individuals older than 1 year into the harvested patch is assumed to be zero. Default model parameter values and uncertainties are derived from statistical model fits to field data (Fagerli et al., 2013, 2015) or based on expert opinion and literature values (Appendix Table A2.2).

The model was configured to represent a patch of area 200 m^2 and was first run for 50 years spin-up without harvesting, starting with zero kelp to produce a steady state urchin barren initial condition. It was then run for a further 20 years with annual harvesting to remove all urchins above a given minimum size. To account for uncertainty the spin-up and harvesting runs were performed as 500-member ensembles with parameter values and stochastic recruitment variability varying between ensemble members following the uncertainty model (Appendix Table A2.2). The harvesting runs were used to explore how the expected harvest yield (as gonad biomass) and its uncertainty varied as a function of the minimum harvested size, assumed to be a fixed constant as a simple management strategy.

Optimal harvesting was investigated first for a present-day scenario, then for a near-term scenario of 0.8°C warming and 100 µatm increase in partial pressure of CO_2 (pCO_2). These physical-chemical changes were based on projections for the period 2030–2050 vs. 2000–2020 under a business-as-usual scenario (SRES A1B) using the SINMOD ocean biogeochemical model (Slagstad et al., 2015). Temperature and pCO_2 sensitivities were parameterized in the kelp growth rate, urchin recruitment flux, and urchin mortality (Appendix Tables A2.1 and A2.2). It is assumed that observed differences in urchin recruitment and mortality between Hammerfest (70.66°N) and Vega (65.67°N) urchin barren sites are associated with bi-decadal mean temperature, with the Vega sites 1.76°C warmer (from bias-corrected SINMOD output, Wallhead et al., 2017). pCO_2 sensitivities were based on literature results from laboratory experiments (see Appendix Table A2.2). Sensitivity of adult urchin growth and mortality to the 100 µatm pCO_2 increase (~0.1 unit decrease in pH) was neglected based on field observations (Uthicke et al., 2016) and experimental results (Siikavuopio et al., 2007; Stumpp et al., 2012; Dupont et al., 2013), noting that the need to interpolate between control conditions and experimental treatments that are much more extreme than +100 µatm pCO_2 introduces some uncertainty. Sensitivity of urchin larval growth rate to the projected acidification was similarly neglected (Chan et al., 2015). Temperature sensitivity of urchin gonad index (ratio of gonad mass to body mass) was based on results from 8-week experiments (Siikavuopio et al., 2006), averaging fractional changes during summer (July–August) and winter (December–January) (Eqn A10), and again using bias-corrected SINMOD output to project seasonal warming (from 8.0°C to 8.8°C for summer and from 6.4°C to 7.2°C for winter).

A2.3 Results and discussion

Model behavior and comparison with observations

The model reproduces the paradigmatic behavior of alternative stable states (May, 1977; Scheffer et al., 2001) with abrupt transitions or ‘regime shifts’ occurring between kelp forest and urchin barren states (Filbee-Dexter and Scheibling, 2014; Ling et al., 2015). When the urchin recruitment flux, which is probably the most naturally variable parameter, is varied over plausible ranges of interannual variations, the system displays strong hysteresis (Figure A2.1). Starting in an urchin barren state, reducing the recruitment flux reduces the total urchin density (blue line) but does not allow recovery of the kelp forest until a low threshold density is reached, after which a regime shift to a kelp-dominated state occurs (the modelled threshold is perhaps too low compared to the 5–10 individuals per m^2 cited by Leinaas and Christie, 1996). Once in the kelp-dominated state, the recruitment flux must be raised to a high level in order to trigger the reverse shift back to the urchin barren state (red line). This behavior occurs even in the absence of an explicitly-resolved third party such as a higher predator (Marzloff et al., 2013) or recruitment facilitator (Baskett and Salomon, 2010).

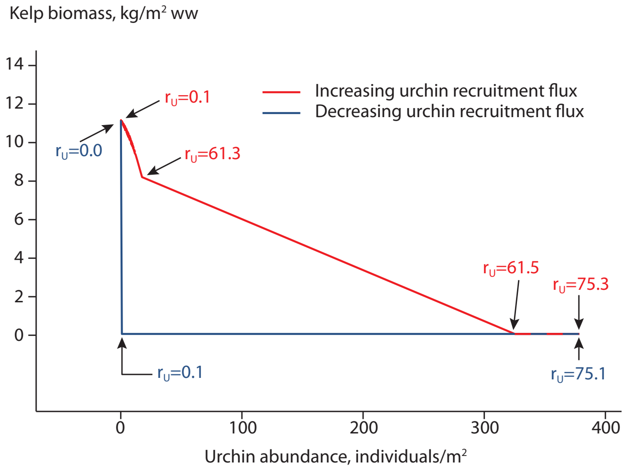

Figure A2.1 Hysteresis plot for the kelp-urchin dynamical model with no stochasticity, varying the parameter r_U (urchin recruitment flux) from 0 to 75.3 and back again to 0.

Comparing with observations from Ling et al. (2015), the model produces a roughly consistent distribution of total abundance density in urchin barrens in northern Norway (Figure A2.2a), allowing for a probable bias in the observations due to undercounting of juveniles (Himmelman, 1986; Russell et al., 1998). This plot provides some independent 'validation', while the remaining comparisons confirm agreement with fitted data. Figure A2.2b shows the exponential decay of abundance over age due to mortality, while Figure A2.2c shows the 'top-heavy' nature of the urchin demographics, with much of the population biomass in the older and less abundant age classes.

Finally, Figure A2.3 shows how the mean size-at-age increases with age, initially at an increasing rate (non von Bertalannfy dynamics). Relative to the Hammerfest barren trajectory, growth is somewhat faster with partial kelp cover (Hammerfest kelp sites). This could not be explained by size-dependent predation alone within the model (not shown), and probably also reflects improved nutrition. Growth in the warmer water (Vega barren) was slightly slower but the same maximum urchin size of around 62 mm is approached in all cases. The effects of kelp presence and warmer water are accounted for in the dynamical model by interpolating between different sets of fitted growth parameters, while the expected shrinking effect of size-dependent harvesting is accounted for by individual-based simulation (see Appendix Table A2.1).

Urchin harvest simulations

In the present-day simulations, the expected annual harvest yield averaged over the 20-year period increases with minimum harvested size until a maximum at around 50 mm is reached, after which it declines (Figure A2.4). Averaging over the first 10 years only, the expected yield is slightly increased and the optimal minimum size is slightly reduced, while the opposite is true if averaging over the last 10 years only. This reflects a decadal population decline and the benefit of looser harvesting restrictions for short-term gains as opposed to the optimal restrictions for 'sustainable' yield. Uncertainty in yield due to parameter uncertainty and stochastic variability is substantial (see error bars). However, this uncertainty has little bearing on the optimal harvesting restriction: to maximize the lower 5%

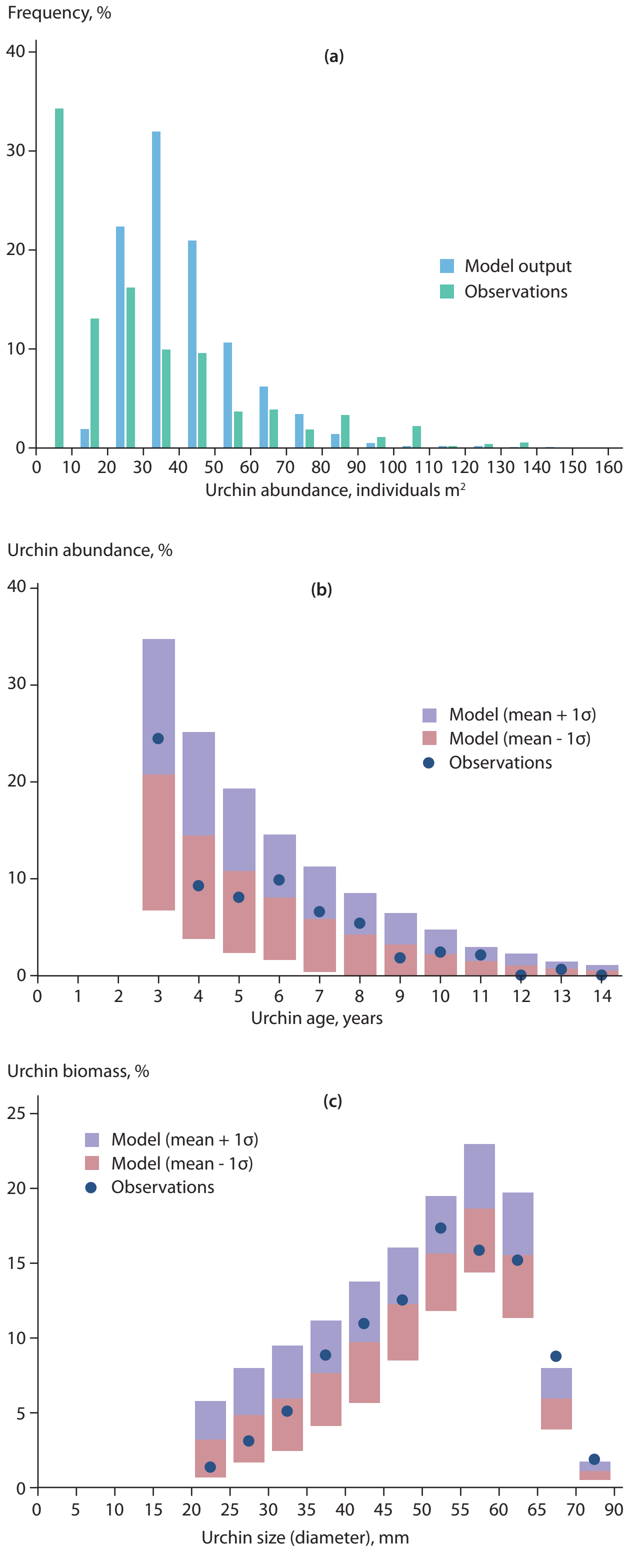

Figure A2.2 The kelp-urchin dynamical model is run 500 times with the default parameter set and stochastic urchin recruitment. Initial conditions are those of an urchin barren. Plot 'a' compares model output with observed data from urchin barrens in Norway north of 68°N (Ling et al., 2015). Each bar shows how frequently an observation or simulation ends in a state with a given urchin abundance range (0–10, 10–20, …, 150–160). Plot 'b' compares model age distribution with observed data from Hammerfest urchin barrens (Fagerli et al., 2015). Only urchins of ages 2+ are considered owing to potential biases in the observations. The graphic shows urchin abundance per one-year age class as a percentage of the total number of urchins of age 2+. Plot 'c' compares model biomass against size-class distribution with observed data from Hammerfest urchin barrens (Fagerli et al., 2015). Only urchins over 20 mm in size are considered owing to potential biases in the observations. The plot shows the amount of urchin biomass per size class as a percentage of the total urchin biomass.

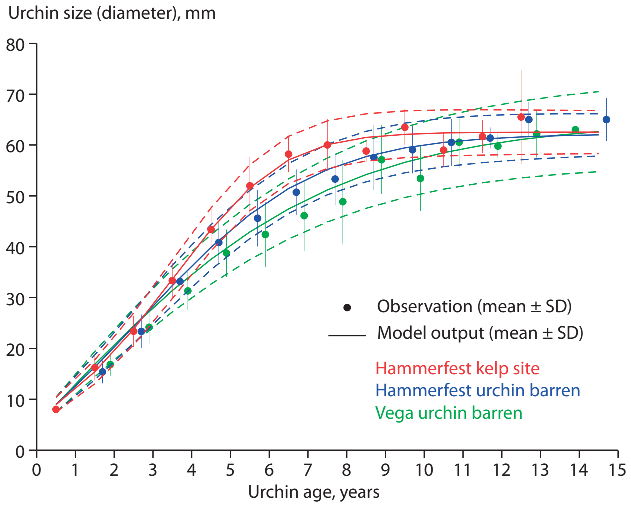

Figure A2.3 Urchin size-at-age relationships in observations from Fagerli et al. (2015) and in the fitted model for the Hammerfest kelp sites, the Hammerfest urchin barren sites, and the Vega urchin barren sites. For clarity, error bars are displaced from age-class central values (0.5, 1.5, …, 14.5 years).

quantile of yield (pessimistic criterion) the optimal minimum size is 45 mm, while to maximize the upper 95% quantile (optimistic criterion) the optimal limit is 50 mm. The optimal limit for any single plausible set of model parameter values and stochastic variability pattern is also well constrained by present knowledge (43–54 mm, 95% CI, not shown). If a limit of 50 mm is applied, the probability of having an urchin barren state, defined as a state with zero kelp, is close to 1 after 20 years.

By contrast, to achieve a high probability (0.8) of provoking a regime shift from an urchin barren to a kelp forest state (defined by kelp biomass ≥0.5 × carrying capacity), it would be necessary to annually cull or remove all urchins ≥10 mm.

In the future simulations, the annual harvest yield is reduced roughly seven-fold due to warming and acidification. However, the optimal harvesting restriction is little affected, remaining close to 45 mm. The culling requirement for a shift to kelp forest with probability 0.8 is slightly relaxed, but remains stringent at around 15 mm.

The apparent robustness of the optimal harvest restrictions may seem surprising, and probably relates to the growth characteristics and mortality rates of the urchins, which are relatively well constrained by data and assumed to be weakly affected by warming and acidification (Figure A2.4; Appendix Tables A2.1 and A2.2). Older urchins are less abundant (because of mortality), but the rate of individual biomass accumulation is maximized at an advanced age because biomass varies roughly as $(\text{size})^3$ and test diameter growth rate only slowly declines with age (Figure A2.3). In addition, gonad index increases with age and body size, further favoring the harvest of larger individuals (gonad mass scales as body mass to the power of 1.47±0.02 in Eqn A10). Harvesting below 50 mm therefore removes many of the most gonad-productive size/age divisions of the population for future years. For comparison, 50 and 51 mm were the minimum legal catch sizes imposed during initial management of the Nova Scotian and Maine green sea urchin fisheries respectively (Miller, 2008; Johnson et al., 2013).

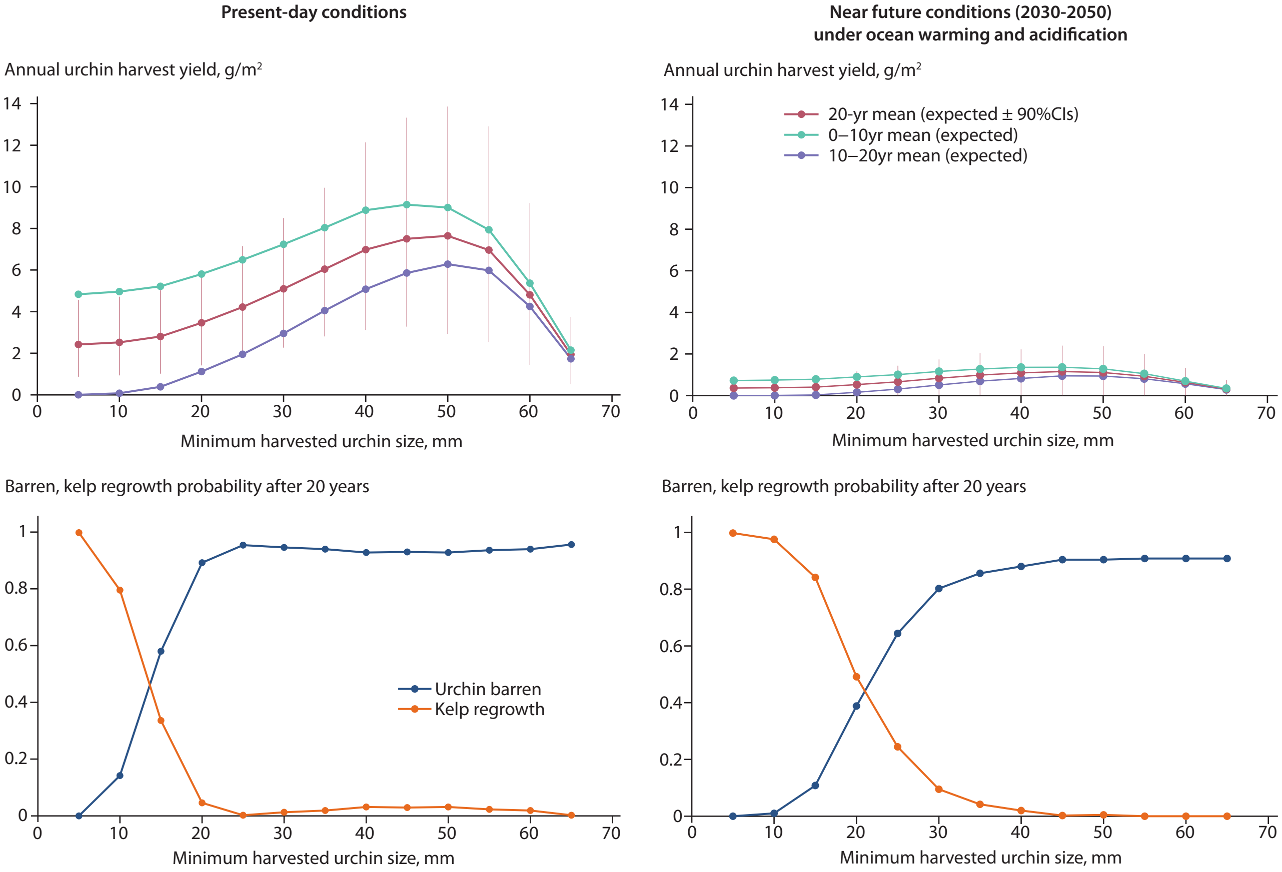

Figure A2.4 Simulated annual urchin harvest yield (gonad biomass per m²) and urchin barren/kelp regrowth probability as a function of the minimum harvested test diameter. Left-hand columns show results for present-day conditions and right-hand columns show the impacts of near-term ocean warming and acidification (2030–2050, SRES A1B, corresponding to +0.8°C, +100 µatm $p\text{CO}_2$ ~ -0.1 pH).

Limitations and recommendations for future work

Population models such as used here (Appendix Table A2.1) have the potential to synthesize information from various sources and propagate individual-scale sensitivities to population-level and economic responses. These models cannot, however, fill basic knowledge gaps regarding individual organism sensitivity. For example, this study assumed log-linear variation to interpolate responses between treatment levels separated by 800 µatm $p\mathrm{CO}_2$ (Appendix Table A2.2). This is clearly open to criticism, and a different interpolating function would yield different responses (e.g., a linear response would give more moderate impacts). Such uncertainties cannot be constrained by purely empirical analysis. More experiments are needed with additional treatments distributed over more moderate levels (regression rather than ANOVA designs). Models of individual-scale physiological response (e.g., DEB models, Jager et al., 2016) are also likely to help in addressing between-replicate variability and other practical limitations of experimental studies.

While ocean acidification had significant impacts on the modelled urchin population through effects on recruitment (10–44% decrease per 100 µatm $p\mathrm{CO}_2$ increase, 95% CI, see $f_{p\mathrm{CO}_2}^{r_U}$ in Appendix Table A2.2), the primary factor driving population decline in the future scenario was warming. Based on analysis of field data at Hammerfest (7.0°C) and Vega (8.7°C), warming by just 0.8°C was assumed to decrease recruitment to the 0–1 age class by 44–87% (see $f_T^{r_U}$ in Appendix Table A2.2). As discussed by Fagerli et al. (2013), the observed differences may be partly due to different levels of micropredation on juveniles at the two sites, which may bias the estimate of temperature sensitivity if this effect does not strongly correlate with warming. However, it is not implausible that this is a robust effect on larval survival and embryonic development as summer temperatures approach the 10°C tolerance limit proposed by Stephens (1972) (Rinde et al., 2014). Such precipitous sensitivities of embryonic development have been mapped experimentally for several other echinoderm species (Karelitz et al., 2017: their figure 4) and it seems high time that the response of *S. droebachiensis* be mapped in a similarly quantitative fashion. In general, longer-term and trans-generational sensitivities should be investigated to take into account effects on the different life-history stages, evolutionary processes and trans-generational plasticity (Calosi et al., 2016).

Improved diet through kelp availability may partially compensate the impacts of warming and acidification on urchin recruitment. In the present model this was accounted for using a statistical relationship derived from field data, suggesting a 17–36% increase in gonad index moving from the ‘barren’ to ‘kelp’ stations (95% CI, Eqn A10, Appendix Table A2.1). Experiments have shown more dramatic effects of diet on green sea urchins, with gonad indices increased by factors of 2–4 relative to wild populations (Meidel and Scheibling, 1999; Garrido and Barber, 2001; Pearce et al., 2004; Siikavuopio et al., 2006). This suggests a potential for stronger compensation in the future climate, although it seems unlikely that natural improvements will be as extreme as observed in experiments where urchins are fed prepared/optimized food in excess.

It should be noted that this model does not account for effects of kelp regrowth on the local seawater $p\mathrm{CO}_2$ or pH, which may act to compensate the increase in $p\mathrm{CO}_2$ within the harvested patch (Krause-Jensen et al., 2015, 2016). However, results from simulations with 0.8°C warming and no acidification effect (not shown) suggest that even if the $p\mathrm{CO}_2$ increase is fully compensated, there will still be a roughly five-fold decrease in the maximum harvest yield due to warming alone, and the requirements for optimal harvesting and culling will be little affected.

Aside from the warming/acidification responses, the model also assumed no net immigration of urchins older than 1 year into the harvested patch. This will not be valid for small patches or for targeted harvesting patches that cover for example the edges of a kelp forest, where individuals are known to aggregate in grazing fronts (Miller, 2008; Johnson et al., 2012, 2013). In such cases, the influx of immigrants, or ‘conveyor belt’, may favor more aggressive harvesting strategies and support higher yields per unit area, and indeed may be a prerequisite for a profitable fishing site (Johnson et al., 2013). The present model could be extended to account for net immigration if suitable estimates of immigration rates/fluxes can be obtained. The model also neglected any correlation between annual recruitment to the 0–1 age class and adult abundance within the harvested patch. This is probably a reasonable assumption for non-retentive systems (Fagerli et al., 2013; Marzloff et al., 2013), but would need revision for other systems. However, it seems unlikely that these effects would significantly alter the projected impacts on yield shown in Figure A2.4, as long as the larvae and juveniles that ultimately source the harvested stock are exposed to ocean warming and acidification.

Finally, it should be noted that the default kelp grazing rate of 4% of body mass per day (g_{us} in Appendix Table A2.2) was based on laboratory experiments with urchins fed in aquaria (Larson et al., 1980; Meidel and Scheibling, 1999) and may overestimate the grazing rate in nature, where urchins must move around more in order to graze the kelp. A lower grazing rate is expected to relax the urchin culling requirement for kelp recovery, but additional simulations did not show strong sensitivity of the results (decreasing g_{us} by a factor of 2 over the parameter ensemble raised the culling threshold to ~12 mm and reduced the optimal harvesting limit to 45 mm in present-day simulations, results not shown).

A2.4 Conclusions

This study developed a simple kelp-urchin dynamical model to investigate the potential impacts of ocean warming and acidification on urchin harvest yield and optimal management strategies in northern Norway. Under changes projected for the next 30 years (+0.8°C, +100 µatm $p\mathrm{CO}_2$) urchin harvest yield underwent a roughly seven-fold decrease, while the optimal minimum test diameter (size limit) of harvested individuals remained around 45–50 mm in both present-day and future simulations. In order to provoke a regime shift from an urchin barren to a kelp forest state with high probability, simulations suggested a need to annually cull or remove all urchins larger than 10 mm, and this requirement was also little affected by warming and acidification. These results should be treated as provisional pending further investigation of organism sensitivities at moderate levels of warming and acidification, of grazing rates in natural populations, and of other ecosystem effects including disease and higher predation. Simplifying assumptions about the harvesting strategy and harvested area, notably that of zero net immigration, may also restrict the validity of present results. On the other hand, the model was able to synthesize information from various sources, including field observations, and was fast to run, enabling an ensemble-based assessment of modeling uncertainty. The model thus appears to be a potentially useful tool for harvest optimization and ecosystem management in the context of a changing climate.

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Appendix: Kelp-urchin model formulation and parameterization

Table A2.1 Parameterizations and variables used in the kelp-urchin dynamical model.

Eqn No.	Module	Formulation	Variables [units]
A1	Kelp dynamics	$\frac{dS}{dt}=r_S+\alpha_{Sc}\left(1-\frac{S}{K_S}\right)S-g_{US}UB_G$ $\alpha_{Sc} = (f_T^{\alpha_S})^{\Delta T/1°\mathrm{C}}(f_{pCO_2}^{\alpha_S})^{\frac{\Delta pCO_2}{100\mu atm}}\alpha_S$ (see note 1)	S = kelp biomass density [kgww/m^2] (see note 1) α_{Sc} = kelp growth rate corrected for (T, pCO$_2$) changes relative to Hammerfest present-day [per year] UB_G = grazing urchin biomass density [kgww/m^2] (see Eqn A3) ΔT = climatic (bi-decadal) change in annual mean temperature [°C] ΔpCO$_2$ = climatic (bi-decadal) change in annual mean pCO2 [μatm]
A2	Urchin dynamics	$U_{i+1,j} = R_{i,(j=1)} + p_{Ui,j-1}U_{i,j-1}$	$U_{i,j}$ = urchin abundance density in year *i*, age class *j* [ind/m^2] $R_{i,j}$ = annual recruitment [ind/m^2] $p_{Ui,j}$ = net annual survival probability
A3	Grazing urchin biomass density in year *i*: UB_{Gi}	$UB_{Gi} = \sum_{k=1}^{n_{sizes}} UBs_{i,k}\gamma_{Gk}$ $UBs_{i,k} = A^{-1}\sum_{inds} UIBM\{s_{k-1} \leq UITD < s_k\}$ $\gamma_{Gk} = \frac{1}{\Delta s_k}\int_{s_{k-1}}^{s_k}\omega_G(r)dr$ $\omega_G(s) = \min\left(1, \max\left(0, 0.5 + \frac{(s-s_{0.5G})}{2ds_{0.5G}}\right)\right)$	$UBs_{i,k}$ = urchin biomass density in year *i*, size class *k* [kgww/m^2] A = patch area [m^2] *UIBM*, *UITD* = urchin individual body mass [kgww] and test diameter [mm] (see Eqns A8, A9) γ_{Gk} = grazing activity factor averaged over size class *k* $\omega_G(s)$ = grazing activity factor (s_{k-1}, s_k) = (minimum, maximum) urchin size in size class *k* [mm]
A4	Urchin annual recruitment in year *i*: R_i	$R_i = f_{Si}^{r_U} f_{Ai}^{r_U} r_{Uc} e^{\varepsilon_{rUi}}$ $f_{Si}^{r_U} = (f_S^{r_U})^{S_i/K_S}$ $f_{Ai}^{r_U} = \max\left(0,1-\pi\sum_j(0.5\times10^{-3}UTDM_{ij})^2U_{ij}\right)$ $r_{Uc} = (f_T^{r_U})^{\Delta T/1°\mathrm{C}}(f_{pCO_2}^{r_U})^{\frac{\Delta pCO_2}{100\mu atm}}r_U$ $\varepsilon_{rUi} \sim N(-0.5\sigma_{rU}^2, \sigma_{rU}^2)$	$f_{Si}^{r_U}$= limitation due to kelp cover $f_{Ai}^{r_U}$ = limitation due to substrate area occupied by sea urchins $UTDM_{ij}$ = mean urchin test diameter in year *i* and age class *j* [mm] (see Eqn A7) r_{Uc} = annual recruitment corrected for (T, pCO$_2$) changes relative to Hammerfest present-day [ind/m^2] ε_{rUi} = stochastic variation in annual recruitment in year *i* (lognormal model)
A5	Urchin net annual survival probability in year *i*, age class *j*: $p_{Ui,j}$	$p_{Ui,j} = (N_{Ui,j} - N_{UHi,j}^{sim} - N_{UMi,j}^{sim})/N_{Ui,j}$ $N_{UHi,j}^{sim} = \{UITD \geq s_H\}_j$ $N_{UMi,j}^{sim} \sim \{\sum_k^{nsizes} B(N_{Ui,k}^{Hcorr}, p_{UMi,k})\}_j$ $N_{Ui,k}^{Hcorr} = \{s_{k-1} \leq UITD < \min(s_k, s_H)\}$	$N_{Ui,j}$ = number of urchins in modelled area (year *i*, age class *j*) $N_{UHi,j}^{sim}$ = simulated number of urchins harvested (year *i*, age class *j*) $N_{UMi,j}^{sim}$ = simulated number of urchin deaths (year *i*, age class *j*) $N_{Ui,k}^{Hcorr}$ = number of urchins in year *i*, size class *k* after removing harvested individuals $p_{UMi,k}$ = urchin (binomial) mortality probability in year *i*, size class *k*
A6	Urchin annual mortality probability in year *i*, size class *k*: $p_{UMi,k}$	$p_{UMi,k} = 1 - p_{Uc}(1 - p_{Pi}\gamma_{Pk})$ $p_{Uc} = (f_T^{p_U})^{\Delta T/1°\mathrm{C}}p_U$ $p_{Pi} = 1 - (f_S^{p_U})^{S_i/K_S}$ $\gamma_{Pk} = \frac{1}{\Delta s_k}\int_{smin_k}^{smax_k}\omega_P(r)dr$ $\omega_P(s) = \min\left(1, \max\left(0, 0.5 - \frac{(s-s_{0.5P})}{2ds_{0.5P}}\right)\right)$	p_{Uc} = barren state survival probability corrected for warming p_{Pi} = kelp-associated predation probability (see note 2) γ_{Pk} = kelp-associated predation susceptibility factor averaged over size class *k* $\omega_p(s)$ = predation susceptibility

A7	(Mean, Standard Deviation) of urchin test diameter in year *i*, age class *j*: ($UTDM_{i,j}$, $UTDSD_{i,j}$)	$UTDM_{i+1,1} = b_0$ $UTDM_{i+1,j>1} = b_{1i} + b_{2i}UTDM_{i,j-1} + b_{3i}(UTDM_{i,j-1})^2 + dUTDM^{sim}_{Hi,j-1}$ $UTDSD_{i+1,1} = c_0$ $UTDSD_{i+1,j>1} = c_{1i} + c_{2i}UTDM_{i,j-1} + dUTDSD^{sim}_{Hi,j-1}$	($dUTDM^{sim}_{Hi,j}$, $dUTDSD^{sim}_{Hi,j}$) = simulated changes in the (means, standard deviations) of urchin test diameters due to harvesting in year *i*, age class *j* [mm] ($b_{1\text{-}3i}$, $c_{1\text{-}2i}$) = parameters of urchin growth dynamics, adjusted for temperature and kelp cover in year *i* (see note 3)
A8	Urchin individual test diameter (*UITD*)	$UITD = UTDM + UTDSD \times \delta_{TD}$ $\delta_{TD} \sim N(0,1)$	δ_{TD} = individual random variation in test diameter within given age class
A9	Urchin individual body mass (*UIBM*)	$\log UIBM = d_0 + d_1 \log UITD + \delta_{BM}$ $\delta_{BM} \sim N(0, \sigma^2_{BM})$ (see note 4)	δ_{BM} = individual random variation in log (body mass) for given test diameter
A10	Urchin individual gonad mass (*UIGM*)	$\log UIGM = e_0 + e_1 \log UIBM + e_2 F_{Diet} + \log f^{GI}_T + \delta_{GM}$ $F_{Diet} = \min(1, \frac{S/KS}{SA_I})$ $f^{GI}_T = 1 + \frac{1}{2}\left(df^{GIs}_T \Delta T_s + df^{GIw}_T \Delta T_w\right)$ $\delta_{GM} \sim N(0, \sigma^2_{GM})$ (see note 4)	F_{Diet} = factor (0–1) accounting for improved diet due to kelp availability f^{GI}_T = adjustment factor for annual urchin gonad index due to warming $df^{GIs/w}_T$ = fractional change in urchin gonad index per 1°C summer/winter warming $\Delta T_{s/w}$ = climatic (bidecadal) change in summer/winter temperature [°C] δ_{GM} = individual random variation in log (gonad mass) for given body mass and diet factor

(1)The differential equation for *S* is solved analytically for constant UB_G to give an annual update between the yearly time steps of the age-structured urchin model. For this calculation the grazing biomass UB_G is averaged between values at the start and end of the annual time step.

(2)Urchin predation is from various sources (notably crabs and ducks, Sivertsen, 2006) and is assumed to increase with kelp cover (Fagerli et al., 2014); here the net effect is modelled implicitly via increased urchin mortality probability.

(3)Parameters (*b*, *c*) were fitted to data from Fagerli et al. (2015) (see Figure A2.3). Base values for the dynamical model were taken from the Hammerfest barren fits. These values were adjusted for warming by linearly interpolating between fitted values for Hammerfest and Vega barrens, assuming 1.76°C temperature difference between the two regions. To adjust for kelp cover, the approach taken was to interpolate between fitted values for Hammerfest kelp and barren sites, assuming kelp cover = 0.5 ± 0.1 at the Hammerfest kelp sites. Adjustments were varied to account for uncertainty in the assumed kelp cover at the Hammerfest sites and were capped to avoid extrapolation. Parameter sets (*b*, *c*) and their adjustments were varied over the parameter ensemble, accounting for covariances via Cholesky decomposition.

(4)Parameters (d, e, σ^2_{BM}, σ^2_{GM}) were fitted to data from Norwegian green sea urchins (Hartvig Christie, NIVA, 03/08/2017 pers. comm.). Fitted equations are: log(UIBM [gww]) = -7.086±0.030 + 2.818±0.008 × log(UITD [mm]), with $\sigma_{BM} = 0.167$, and log(UIGM [gww]) = -5.037±0.057 + 1.472±0.020 × log(UIBD [gww]) + 0.266±0.050 × F_{Diet}, with $\sigma_{GM} = 0.980$. Parameters (*d*, *e*) were varied over the parameter ensemble, accounting for covariances via Cholesky decomposition. Uncertainty in the diet factor due to uncertainty in the assumed level of kelp cover (biomass as a fraction of carrying capacity) at the 'kelp' field station in Indreskjær ($SA_I = 0.5 \pm 0.1$) was accounted for by varying SA_I over the parameter ensemble.

Table A2.2 Parameters used in kelp-urchin dynamical model, showing default values, uncertainty parameters assuming lognormal [LN] or normal [N] distributions, 95% credibility intervals, and basis references/methods. Ensemble parameter values are random sampled from (log)normal distributions with medians defined by the default values and (logarithmic) standard deviations defined by the uncertainty parameters (σ). Sensitivity factors are referenced to present-day climate in a Hammerfest green sea urchin barren.

θ	Meaning [units]	Default value	σ [LN/N]	95% CI	Basis$^{(\mathrm{Note})}$
r_s	Kelp recruitment [gww/m^2/year]	25	1.0 [LN]	4 - 180	Marzloff et al. (2013)
α_s	Kelp growth rate [year^{-1}]	1.0	0.5 [LN]	0.4 - 2.7	Leinaas and Christie (1996)
$f^{\alpha_s}_{T}$	Sensitivity factor for kelp growth rate per 1°C warming	1.068	0.009 [LN]	1.05 - 1.09	Average of logarithmic gradients from Olischläger et al. (2017) and Iñiguez et al. (2016)
$f^{\alpha_s}_{pCO_2}$	Sensitivity factor for kelp growth rate per 100 µatm $p\mathrm{CO}_2$ increase	1.014	0.017 [LN]	0.98 - 1.05	Average of logarithmic gradients from Olischläger et al. (2017) and Iñiguez et al. (2016)$^{(1)}$
K_S	Annual kelp carrying capacity [kgww/m^2]	11	0.2 [LN]	7 - 16	Hartvig Christie, NIVA, 17/08/2017 pers. comm.
g_{US}	Urchin grazing rate [gww kelp/gww urchin/year]	14.6	0.2 [LN]	10 - 22	Larson et al. (1980); Meidel and Scheibling (1999)
$s_{0.5G}$	Urchin test diameter for half maximum grazing activity [mm]	20	2 [N]	16 - 24	Camilla Fagerli, NIVA, 14/11/2017 pers. comm.
$ds_{0.5G}$	Change in test diameter to reach half maximum grazing activity [mm]	5	0.4 [LN]	2 - 11	Camilla Fagerli, NIVA, 14/11/2017 pers. comm.
r_U	Urchin mean annual recruitment into 0–1 year age class [ind/m^2]	12.5	0.25 [LN]	8 - 20	(observed mean abundance in Hammerfest barren) × (expected fraction in 0–1 year age class)$^{(2)}$
σ_{r_U}	Logarithmic standard deviation in urchin annual recruitment	0.82	0.15 [LN]	0.61 - 1.10	GLMM fit to age-structured data from Fagerli et al. (2015)$^{(3)}$
$f^{r_U}_{S}$	Sensitivity factor for urchin recruitment at maximum kelp cover	0.025	1.10 [LN]	0.003 - 0.22	GLMM fit to juvenile abundance data from Fagerli et al. (2013)$^{(4)}$
$f^{r_U}_{T}$	Sensitivity factor for urchin recruitment per 1°C warming	0.20	0.47 [LN]	0.08 - 0.48	GLMM fit to juvenile abundance data from Fagerli et al. (2013)$^{(4)}$
$f^{r_U}_{pCO_2}$	Sensitivity factor for urchin recruitment per 100 µatm $p\mathrm{CO}_2$ increase	0.71	0.12 [LN]	0.56 - 0.90	Combined sensitivity of egg production, fertilization success, and larval/juvenile survival (Bögner et al., 2014; Dupont et al., 2013) $^{(5)}$
p_U	Urchin annual survival in reference urchin barren	0.71	0.047 [LN]	0.65 - 0.78	GLMM fit to age-structured data from Fagerli et al. (2015)$^{(3)}$
$f^{p_U}_{S}$	Sensitivity factor for urchin survival at maximum kelp cover	0.72	0.12 [N]	0.47 - 0.96	GLMM fit to age-structured data from Fagerli et al. (2015)$^{(3)}$
$f^{p_U}_{T}$	Sensitivity factor for urchin survival per 1°C warming	0.95	0.038 [LN]	0.88 - 1.02	GLMM fit to age-structured data from Fagerli et al. (2015)$^{(3)}$
$s_{0.5P}$	Urchin test diameter for half maximum predation susceptibility [mm]	50	5 [N]	40 - 60	Camilla Fagerli, NIVA, 08/11 pers. comm.
$ds_{0.5P}$	Change in test diameter to reach half maximum predation susceptibility [mm]	5	0.4 [LN]	2 - 11	Camilla Fagerli, NIVA, 08/11/2017 pers. comm.
df^{GIs}_{T}	Fractional increase in urchin gonad index per 1°C summertime warming	0.17	0.055 [N]	0.06 - 0.27	Siikavuopio et al. (2006)
df^{GIw}_{T}	Fractional increase in urchin gonad index per 1°C wintertime warming	0.05	0.028 [N]	-0.01 - 0.10	Siikavuopio et al. (2006)

Table notes on following page

(1)Logarithmic gradients were first averaged over temperature treatments within each study; any interaction effects were small compared to the variation in $p\mathrm{CO}_2$ effect between studies.

(2)Based on data from Fagerli et al. (2015), with uncertainty estimated by Monte Carlo simulation, accounting for error in the observed mean abundance and in the model estimates of the expected fraction (dependent on parameter p_U). The resulting correlation in estimates of p_U and r_U is accounted for in the parameter ensemble.

(3)Assuming Poisson sampling errors and lognormal recruitment variability, the observed counts in different age classes should approximately follow a Poisson-lognormal distribution under the dynamical model, with mortality causing a log-linear variation over age (this neglects mortality rate variations due to varying kelp biomass). This is a type of Generalized Linear Mixed-effects Model (GLMM) with Poisson family and log link function. The following function in R was fitted using package 'lme4' (Bates et al., 2015): log(count) ~ Age + Kelp_cover + ΔT+ Age:Kelp_cover + Age:ΔT + (1|Station/Age), where Age is the urchin age in years, Kelp_cover is the kelp biomass as a fraction of carrying capacity (assumed to be 0.5/1.0 in Hammerfest/Vega kelp sites and 0 in urchin barren sites) and ΔT is the difference in bi-decadal average bottom water temperature relative to Hammerfest (=1.76°C for Vega sites, 0°C for Hammerfest sites). This model assumes independent lognormal spatial variability between sampling stations and independent lognormal temporal variability between ages (years) within stations. Counts for ages <2 years were excluded due to likely sampling biases, and counts from Vega kelp forest sites were excluded due to excessive zeros. The fitted coefficients for (Age, Age:Kelp_cover, Age:ΔT) are used to constrain (p_U, f_S^{pU}, f_T^{pU}) in the dynamical model while the other fixed effects are used only to improve model fit since they are probably impacted by variations in sampling effort between stations. The fitted random effect variances were used to place conservative constraints on σr_U, and the covariance matrix of the fixed effects was used to produce covarying ensemble member parameters.

(4)The following Poisson-lognormal GLMM (see note 3) was fitted to the juvenile (<2 cm) urchin abundance per m^2 data from Fagerli et al. (2013): log(count) ~ Kelp_cover + ΔT + (1|sample), hence assuming independent lognormal variability between each abundance sample.

(5)It is assumed that $r_U = Ep_Fp_Lp_J$ where E is the egg production, p_F is the probability of successful fertilization, and $p_{L,J}$ are conditional probabilities of survival through larval and juvenile stages. Hence the net sensitivity factor is a product of the sensitivities of each process/stage. The 100 µatm sensitivity factors for (E, p_L, p_J) were estimated from Dupont et al. (2013) using only results from acclimated organisms and log-linearly interpolating over the 800 µatm treatment level, hence for example $f^{E}_{pCO_2} = (E(1200\mu\mathrm{atm})/E(400\mu\mathrm{atm}))^{1/8}$. For p_F a log-linear model was fitted to the data up to 1300 µatm from Bögner et al. (2014) and again interpolated back to 100 µatm change. Uncertainties were calculated by Monte Carlo simulation.

Annex 3

Ocean services for the Barents Sea – socio-economic effects of ocean warming and acidification on the Northeast Arctic cod fishery

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Contents

A3.1 Introduction

Marine fisheries play a central role in world food supply. Fish provides at least 15% of per capita animal protein intake for 4.5 billion people (Béné et al., 2015). Worldwide roughly 500 million people are directly dependent on fisheries for their livelihoods (FAO, 2014). Ocean warming and in recent years also ocean acidification have been identified as potential major stressors to marine fisheries, threatening the sustainable use of these renewable resources and their associated socio-economic systems. This case study examines the socio-economic effects of ocean warming and acidification on Northeast Arctic cod (*Gadus morhua*), which forms the basis of a commercially important fishery in a region expected to continue to experience strong environmental change.

Ocean warming affects species distribution (Perry et al., 2005; Rijnsdorp et al., 2009; Last et al., 2011) and vital rates, such as growth and mortality (Björnsson et al., 2007). Moreover, the effect of carbon dioxide (CO_2) dissolution in upper ocean waters, termed ocean acidification, has been identified as an additional stressor for marine fish stocks. Its effects are system- and species-specific. Experimental work shows effects on behavior (Simpson et al., 2011; Devine et al., 2012; Nilsson et al., 2012; Pimentel et al., 2016) and vital rates (Baumann et al., 2012; Pimentel et al., 2016; Stiasny et al., 2016). Although evidence for commercially exploited fish species is still sparse, it does suggest non-uniform reactions for different species or stocks (Frommel et al., 2012, 2013, 2014, 2016; Maneja et al., 2012; Stiasny et al., 2016).

In the North-Atlantic region, cod has for centuries been a major natural resource, and has even been described as a 'fish that changed the world' (Kurlansky, 1997). The impact of temperature variations on recruitment success and hence stock dynamics of North-Atlantic cod has long been recognized (e.g., Planque and Frédou, 1999). While some cod stocks might react positively to ocean warming, most stocks in the eastern North-Atlantic are negatively affected by temperature increase as recruitment success is lowered (Drinkwater, 2005). The impacts of ocean acidification have been studied in laboratory experiments on Norwegian coastal cod (Frommel et al., 2012) and western Baltic cod (Stiasny et al., 2016), which showed increased larval mortality under projected end-of century business-as-usual conditions, which might have severe consequences for population dynamics (Voss et al., 2015).

An important and rarely considered issue is how the combined effects of ocean warming and acidification will play out for commercially important fisheries.

This case study focuses on Northeast Arctic cod, because the Arctic Ocean and the marginal Barents Sea are considered

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* In this annex the term 'Northeast Arctic cod' is used to denote the Atlantic cod (*Gadus morhua*) population in the Barents Sea area.

an 'early warning system' for climate change (IPCC, 2014). Strong sea-ice reduction due to ocean warming has led to a decrease in albedo and consequently more solar radiation is absorbed by the surface waters, which causes ocean temperature to rise even faster. Owing to this positive surface albedo feedback as well as to temperature feedbacks (Screen and Simmonds, 2010; Pithan and Mauritsen, 2014), the Arctic is warming faster than the global average, a phenomenon known as 'polar amplification' (IPCC, 2014). The Arctic Ocean is also projected to experience the strongest acidification of the global ocean (Steinacher et al., 2009) with models projecting undersaturation of surface waters with respect to aragonite (a calcium carbonate mineral) in the Arctic by 2050 (Anderson et al., 2010), subject to high regional variability (Popova et al., 2014). Indeed observations already repeatedly show undersaturation of Arctic surface waters (Chierici and Fransson, 2009; Yamamoto-Kawai et al., 2009; Chierici et al., 2011; Mathis et al., 2012).

In this case study the effect of changing ocean temperature and pH on Northeast Arctic cod recruitment is quantified using time-series data on ocean temperature from the Kola Section – a well-known indicator of Barents sea climate (Boitsov et al., 2012) – as well as experimental data from Stiasny et al. (2016) that show acidification to have a strongly negative effect on recruitment. This is followed by an investigation of how best to adapt the management of the Northeast Arctic cod fishery to changing environmental conditions using an ecological-economic optimization model. The model is run accounting for the ocean warming effect only or the combined effects of ocean warming and acidification. It quantifies impacts on the Northeast Arctic cod fishery in terms of ecological (stock size), economic (catches, profits), and social (fishing mortality as a proxy for employment) terms.

A3.2 Northeast Arctic cod under ocean warming and acidification

The Northeast Arctic cod (ICES Subareas I and II) has a long fishing history. Catch levels in the mid-1970s were about 900,000 tonnes. After that they showed periods of increasing and decreasing catches (ICES, 2016). The lowest recorded catch in the post-war period was about 212,000 t in 1990, which was followed by a rapid increase in catches until they stabilized at around 750,000 t from 1994 to 1997. Catches decreased again until 2000 (414,000 t) but were above the long-term average between 2010 and 2015. Catches are currently as high as in the mid-1970s.

Northeast Arctic cod spawns in areas along the Norwegian coast at depths of 50 to 200 m. Larvae and juveniles migrate northward throughout spring and summer and the juvenile distribution spans most of the southeast Barents Sea in autumn (Stige et al., 2015). Historically, the nursery grounds (1–3 years), and adult wintering and feeding grounds have shown a similar distribution along the southern Barents Sea and up to Spitsbergen (Sundby, 2000; Vikebø et al., 2005). However, there was a recent shift in cod distribution with cod catches shifting to the northern-most edge of the Barents Sea between 2007 and 2012 (Kjesbu et al., 2014) (see Figure A3.1).

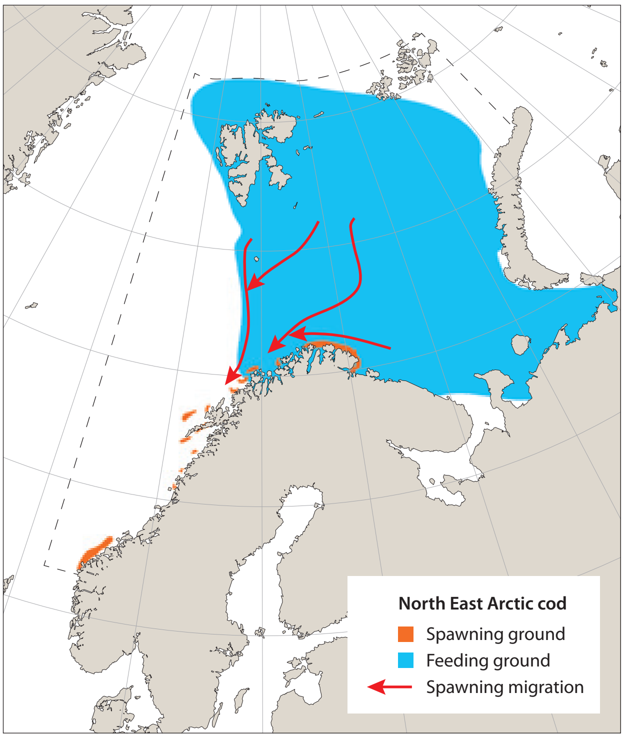

Figure A3.1 Current distribution of the Northeast Arctic cod stock in the Barents Sea.

Temperature effects on cod recruitment have long been explored and their importance is widely recognized (Ottersen et al., 1994; Sundby and Nakken, 2008). However, while the effect of rising temperatures on Northeast Arctic cod recruitment has so far been positive (Kjesbu et al., 2014), which is likely to have supported stock management and assisted in stock recovery over recent years, comparisons between cod stocks suggest that at bottom temperatures below 5°C recruitment increases with increasing sea surface temperature and at bottom temperatures above 8.5°C recruitment decreases (Planque and Fredou, 1999; Drinkwater, 2005).

Ocean acidification is a new area of research and the effects are far less understood. Nevertheless, there is evidence that ocean acidification may have a significant effect on recruitment of Northeast Arctic cod (Stiasny et al., 2016; Koenigstein et al., 2017). It is therefore becoming increasingly important to understand the effect of ocean acidification and how it interacts with the effects of warming.

A3.3 Materials and methods

Population level effects of ocean warming and acidification

Considering the potential impact of ocean warming and acidification on fisheries requires scaling from physiological responses to population-level processes. A first step is to consider how both stressors could modify the parameters of growth, mortality and reproduction. This study concentrates on the modification of the parameters of a Ricker type stock-recruitment relationship (Ricker, 1954) in an age-structured fishery model.

Table A3.1 Estimates for coefficients of stock-recruitment model. $R^2=0.39$, F-statistic:12.97, p-value <0.001.

Parameter	Estimate	Lower bound (5%)	Upper bound (95%)	Unit
φ_0	-7.59	-14.04	-1.13	
φ_1	0.0014	0.0009	0.0019	(1000 tonnes) $^{-1}$
φ_T	4.03	0.74	7.32	Celsius^{-1}
φ_{T2}	-0.45	-0.86	-0.037	Celsius^{-2}

Ocean warming and acidification both cause changes in early life stage mortality rates leading to the density-independent mortality rate, a. In the baseline scenario (warming only) $a=0$, while in the scenarios with warming and acidification, e^{-a} is the fraction of larvae surviving the combined effects of both stressors.

Mortality during the recruitment process is due to density-independent and density-dependent effects, both of which may potentially be affected by warming and acidification. This study follows the approach of Hjermann et al. (2007) and Röckmann et al. (2007) and assumes that the effects of both stressors will only influence density-independent mortality during the recruitment phase. Although Myers and Cadigan (1993) and Fromentin et al. (2001) showed that density-dependent mortality can be important for cod populations in general, there is no evidence linking density-dependent mortality to changing environmental conditions.

Information on the magnitude of ocean acidification effects is based on published experimental work (Stiasny et al., 2016). The resulting modified stock recruitment relationships were applied in an ecological-economic optimization model to quantify the potential environmental effects on the fishery.

Ocean warming effects

To include the effects of ocean warming in the stock-recruitment model for Northeast Arctic cod, this study uses annual time-series data on spawning stock biomass and recruitment numbers supplied by the International Council for the Exploration of the Sea (ICES, 2016). The monthly temperature data were taken from the Kola Section (Boitsov et al., 2012). Temperature data are the monthly integrated mean for 0–200 m depth along the 33°30'E meridian from 69°30 to 77°00'N for each year between 1921 and 2013. Although these data are not depth-specific, the monthly integrated mean is a widely used indicator for the impacts of ocean warming on cod populations in the Barents Sea (Ottersen et al., 2000; Hjermann et al., 2007; Bogstad et al., 2013; Ottersen et al., 2014). Using Ordinary Least Squares regression, it was found that the January temperature statistically best explains recruitment within this time frame. As recruitment data are only available from 1946 onwards, this study uses an annual time series from 1946 to 2013.

Using R_{t+3} for the number of recruits in year t at age 3 years, SSB_t for spawning stock biomass in year t, T_t for the Kola Section in January of year t, and e_t to denote an independent and identically distributed error term, the following stock-recruitment function is estimated:

$$\log(R_{t+3}/SSB_{t-1}) = \varphi_0 - \varphi_1 SSB_{t-1} + \varphi_T T_t + \varphi_{T2} T_t^2 + \varepsilon_t$$

The final estimators are given in Table A3.1.

Ocean acidification effects

Estimates of mortality increase due to ocean acidification are based on experiments by Stiasny et al. (2016) in which the survival of cod larvae was quantified in direct response to increased partial pressure of CO_2 (pCO_2) levels. Eggs and larvae from the Northeast Arctic cod stock caught in the Barents Sea were kept under control conditions (~400–500 μatm corresponding to a pH of 7.6±0.03) and elevated CO_2 conditions (~1100 μatm corresponding to a pH of 7.9±0.15) until 25 days post-hatching, and survival was monitored closely. Projected end-of-century levels of ocean acidification (~1100 μatm under the IPCC RCP8.5 scenario) resulted in a doubling of daily mortality rates compared to present-day CO_2 concentrations during the first 25 days post hatching, a critical phase for population recruitment. The results were consistent under different feeding regimes. This study uses the estimate of Stiasny et al. (2016) in their more conservative scenario that recruitment under end-of-century acidification may be reduced to about a quarter of the levels under baseline CO_2 levels, that is, $e^{-a}=0.245$.

Ecological-economic optimization model

This study used an ecological-economic optimization model for a fishery of an age-structured fish stock with 13 age classes, parameterized for the Northeast Arctic cod fishery, following the modeling approach of Tahvonen et al. (2018). The model calculates the economic optimal fishing effort and related total allowable catch (TAC) to be set under steady state conditions.

$X_{s,t}$ is used to denote the number of fish in age class $s=1, \ldots n$ at the beginning of year t. The number of fish that are caught from age class s in year t is denoted by h_{st}. The study uses n=13 age classes with the last age class subsuming all individual fish that are 13 years or older. Spawning stock biomass follows as

$$\mathrm{SSB}_t = \sum_{s=1}^{n} w_s\, g_s\, x_{st}$$

for which weights-at-age s w_s are taken from ICES (2016, their table 3.8) and maturity-at-age s g_s from ICES (2016, their table 3.11). The relationship between spawning stock biomass and recruitment three years later is captured by calculating recruitment as one-year old fish that then face zero natural mortality until becoming three-year old fish. Because of this implementation strategy, the Ricker stock-recruitment function in the programming code reads (still using the parameter estimates as in Table A3.1, but with the new variable x for recruitment):

$$x_{1,t+1} = \mathrm{SSB}_t e^{\varphi_0 - \varphi_1 \mathrm{SSB}_t + \varphi_T T_t + \varphi_{T^2} T_t^2}$$

The dynamics of the older age classes s=2, …, n follow as

$$x_{s+1,t+1}=\alpha_s\left(1-q_s q_s \ \frac{H_t}{B_t}\right)x_{st}, \qquad s=1, \ldots, n-2$$

$$x_{n,t+1}=\alpha_{n\text{-}1}\left(1-q_{n\text{-}1}(\sigma_t)\,\frac{H_t}{B_t}\right)x_{n\text{-}1,t}+\alpha_n\left(1-q_n(\sigma_t)\,\frac{H_t}{B_t}\right)x_{n,t}$$

where H_t denotes total biomass caught (across all age classes),

$$H_t=\sum_{s=1}^{n} w_s\,h_{st} \ \text{ and } \ B_t=\sum_{s=1}^{n} w_s\,q_s(\sigma_t)x_{st}$$

denotes the efficient stock size (Tahvonen et al., 2018), that is the total fishable biomass when sorting grid spacing σ_t may limit the age-specific catchabilities $q_s(\sigma_t)$ to values below one. The sorting grid became mandatory in the Northeast Arctic in 1997. All fishing trawlers must use a grid with a minimum bar spacing of 55 mm (Sistiaga et al., 2008). The equality

$$h_{st}=q_s(\sigma_t)\,\frac{H_t}{B_t}\,x_{st}$$

implicitly used above follows from the assumption that the share of age class *s* in a catch equals its share in efficient biomass,

$$\frac{w_s\,h_{st}}{H_t}=\frac{w_s q_s(\sigma_t)x_{st}}{B_t}$$

The general functional relationship between sorting grid spacing σ_t (in mm) and the age-specific catchability coefficients $q_s(\theta_t)$ are as found in (Madsen, 2007):

$$q_s(l_s)=\frac{e^{a+bl_s}}{1+e^{a+bl_s}} \quad \text{with} \quad L_{50}=-\ \frac{a}{b}, \ \text{SR}=\frac{\log 9}{b}$$

for which Sistiaga (2008) have estimated the following relationships for NEAC,

$$L_{50}=0.8512\sigma_t+6.0237$$

$$\text{SR}=4.2079e^{\,0.0125\sigma_t}$$

The length-at-age values l_s used above are the means of the values reported by ICES (2016, their tables A5, A7 and A1).

Following the approach of Diekert et al. (2010a,b) and Diekert (2013), this study considers age- and year-specific average landing prices p_s, calculated from the landing tickets of Norwegian trawlers. For each weight category and year, the cumulated catch value in NOK was divided by the respective cumulated live weight caught. To be able to estimate the cost function given the fishing technology and stock size, annual means of the profit margins of Norwegian trawlers, defined as

$$\pi_t\equiv\left(\frac{\text{Profit}}{\text{Revenue}}\right)_t$$

were calculated from Fiskeridirektoratet (2008–2013) data.

Adopting the standard formulation

$$\Pi=pH-\frac{cH}{X^{\gamma}}$$

from biomass models, fishing profits and hence profit margins can be written as

$$\Pi_t=\sum_{s=1}^{n} p_s w_s h_{st}-c_0\frac{H_t}{B_t^{\gamma}}$$

$$\pi_t=\frac{\sum_{s=1}^{n} p_s w_s q_s(\sigma_t)\frac{H_t}{B_t}x_{st}-c_0\frac{H_t}{B_t^{\gamma}}}{\sum_{s=1}^{n} p_s w_s q_s(\sigma_t)\frac{H_t}{B_t}x_{st}}$$

yielding the following equation whose log form can be used to estimate c_0 and γ:

$$c_0 B_t^{1\text{-}\gamma}=(1-\pi_t)\sum_{s=1}^{n} p_s w_s q_s(\sigma_t)x_{st}$$

The estimates of $\ln c_0 = 2.8463$ (1.1514) and $1 - \gamma = 0.8959$ (0.1796) were computed using an Ordinary Least Squares regression with seven data points between 2004 and 2010 (standard errors in parentheses, $R^2 = 0.8327$). Transforming these estimates yielded $c_0 = 17.224$ and $\gamma = 0.104$.

This analysis considers maximum economic yield management, i.e. catch, gear choice and age-specific stock numbers, so as to maximize steady-state economic surplus.

The ecological-economic optimization model is implemented in the AMPL programming language (Fourer et al., 2003) and solved using Knitro 10.3. Programming codes that list all remaining age-specific parameter values are available on request by the authors of this study.

A3.4 Results

Combined effects of fishing, warming and acidification

The first step of the analysis was to study the combined effects of fishing, warming and acidification on the Northeast Arctic cod stock. This was achieved by varying fishing mortality *F* and January temperature *T* to study the combined effects of fishing and warming on the expected stock size in steady state both for the case without and with acidification-induced mortality of early life history stages of cod. In all computations, the sorting grid size was kept at 55 mm, according to current regulations (Sistiaga et al., 2008). The resulting steady-state stock size was computed under the combined stressors of ocean warming and acidification

$$\overline{SSB}_{OAW}$$

and divided by the unfished steady-state biomass for the optimal temperature

$$\overline{SSB}_{T^*}|_{F=0}$$

thereby generating an indicator of how the combined stressors of fishing and warming (Figure A3.2, left panel) – and acidification (Figure A3.2, right panel) – could reduce stock size. One minus this indicator, i.e.

$$1-\frac{\overline{SSB}_{OAW}}{\overline{SSB}_{T^*}|_{F=0}}$$

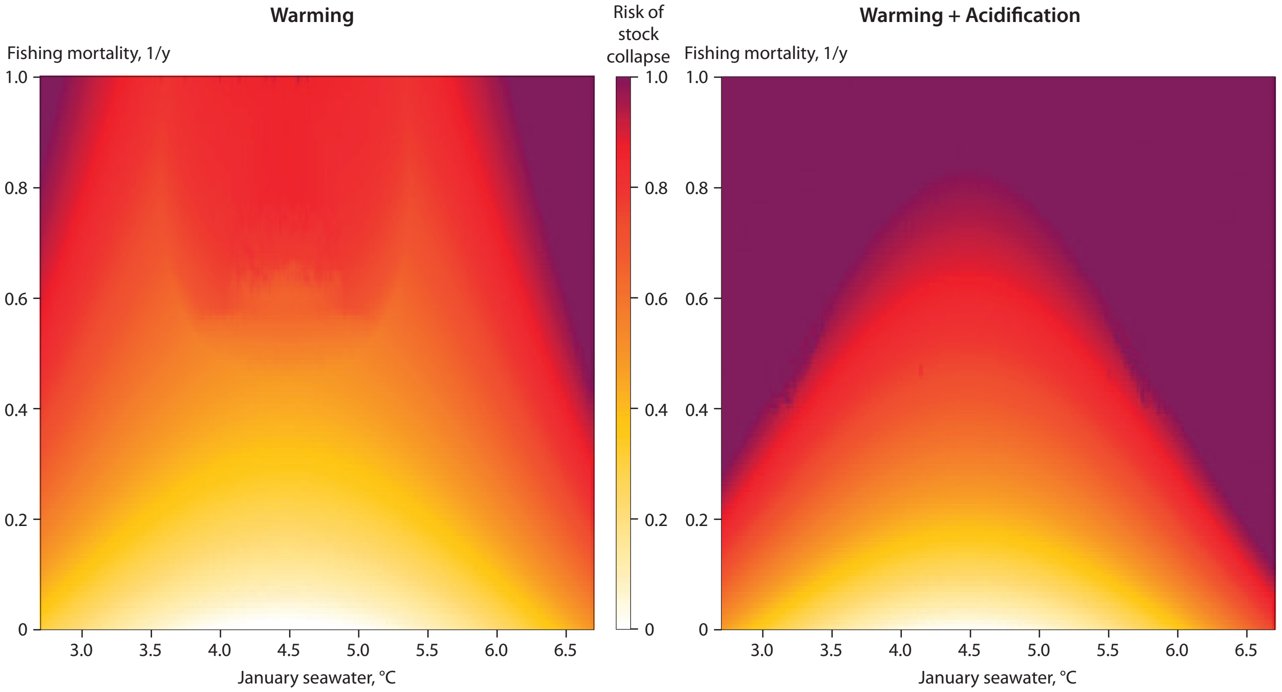

Figure A3.2 'Burning ember' plots showing the combined effect of fishing and warming on the Northeast Arctic cod stock without taking into account extra mortality due to acidification (left panel) and the combined effect of fishing and warming plus the extra mortality due to acidification (right panel). In both plots, the January temperature ranges from 2.7°C (the minimum in the time series) to 6.7°C (somewhat above the maximum of the historical time series) and fishing mortality ranges from zero to one. Color codes show the risk-of-collapse-indicator, with darker shades indicating a higher risk of stock collapse.

was used as an indicator of the risk of stock collapse. Results are shown in Figure A3.2.

Overall, ocean acidification greatly increases the risk of stock collapse. Under the combined effect of warming and acidification, the levels of fishing mortality that the stock can support are much lower than under warming only. An 'optimal' temperature of 4.47°C is roughly visible. At this temperature, the stock could support a much higher fishing mortality for the same risk of collapse. Without acidification effects, even at a fishing mortality of *F=0.5*, the steady-state population size is about half that of the unfished size.

Socio-ecological effects of temperature increase

The second step of the analysis was to calculate the economically optimal fishing management in terms of fishing mortality and gear choice for different levels of temperature increase, and with and without ocean acidification effects. This was done using the age-structured ecological-economic optimization model. The model output illustrates not only optimal fishing mortality (which can be seen as a proxy for employment possibilities), but includes optimal size of the spawning stock, catch level, and profits for the fishery. The temperature range considered again spans 2.7°C to 6.7°C January temperature. The number of recruits is at a maximum of 1100 million for a temperature optimum around 4.5°C. At about 0.9 million tonnes, the corresponding catch is close to the levels reached in the early 1970s. This optimal temperature maximizes net revenues at around NOK 2300 million per year while the spawning stock biomass is kept at an optimal level of around 1.5 million tonnes. The optimal fishing mortality can be interpreted as a proxy for employment possibilities in the fishery because fishing mortality is mainly driven by fishing effort. A high fishing effort is only possible if enough fishermen are employed in the industry. At the optimal temperature of 4.5°C fishing effort reaches a maximum of 0.25 per year. The sorting grid size is chosen such that the mean age of catch of 6.5 years is close to the current level.

Combined effects of ocean warming and acidification

The final step in the analysis was to consider the combined effect of both stressors. Estimates of mortality increase due to ocean acidification are based on experiments in which the survival of Northeast Arctic cod larvae was quantified in direct response to increased pCO_2 levels. End-of-century levels of ocean acidification (~1100 µatm under the IPCC RCP8.5 scenario) resulted in a doubling of daily mortality rates compared to present-day CO_2 concentrations (~400–500 µatm) during the first 25 days post hatching, a critical phase for population recruitment (Stiasny et al., 2016). Applying the effects of both stressors in the recruitment function results in severe outcomes for the fishery (see Figure A3.3). Recruitment and catch under the combined effect of both stressors are reduced to about a quarter of the numbers reached when acidification is excluded. The optimal spawning stock biomass under the optimal temperature for the fishery (4.5°C) is, however, higher under the combined case than under warming only. Spawning stock biomass is kept high in the combined case in order to buffer the adverse effects of ocean acidification. For temperatures both higher and lower than the optimum (4.5°C), spawning stock biomass under both stressors decreases much faster compared to the case without acidification. Hence, under ocean acidification the fishery is much more vulnerable to temperatures that deviate from the optimum. Net revenues decrease to about 10% of those attained without acidification. The optimal fishing mortality is also reduced by about 80%, translating into severe consequences for employment possibilities in the fishery.

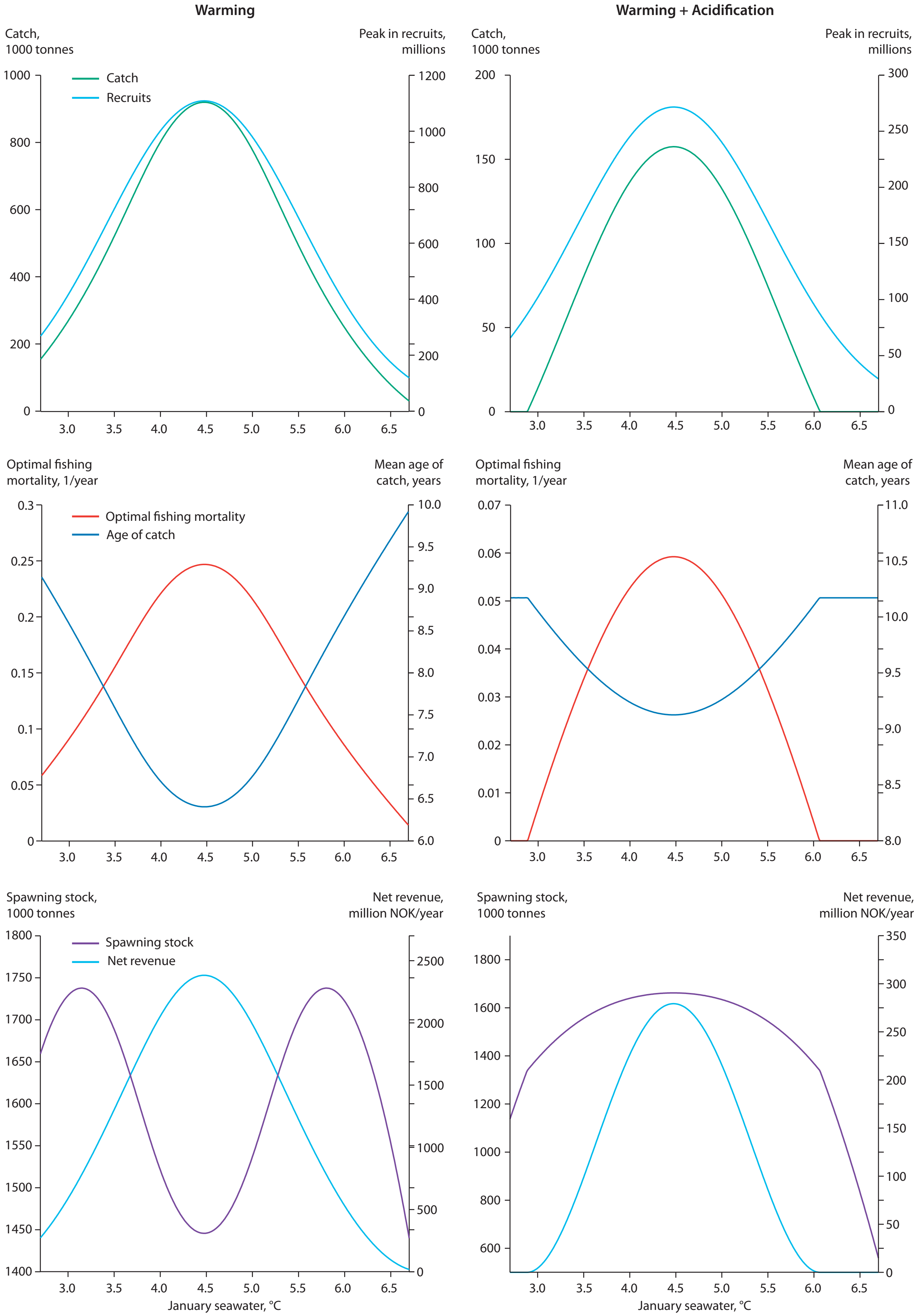

Figure A3.3 Temperature effects on the optimal management of Northeast Arctic cod (*Gadus morhua*) with and without the effect of acidification. Depicted are catch and recruits (top panel), fishing mortality and mean age of catch under optimal management (middle panel), and net revenue and spawning stock biomass (lower panel). The left-hand column show results for warming only, while the right-hand column show results for the combined effect of warming and acidification. It is important to note the marked change in scale for the two sets of vertical axes.

A3.5 Discussion and conclusions

This study used an ecological-economic optimization model of the Northeast Arctic cod fishery to study the biological and economic effects of warming and acidification on one of the commercially important fish stocks in the North Atlantic; a natural environment that can be deemed an 'early warning system' for climate change in marine environments. The analysis examined how increasing temperature and CO_2 affect ecological (stock size), economic (profits), consumer-related (harvest) and social (fishing effort) indicators, ranging from present-day conditions to future climate change scenarios. Results show that near-term climate change will benefit the fishery, because temperature is still (slightly) below the optimum for cod reproduction. However, under a business-as-usual scenario of warming and acidification this large fishery may be at risk of collapse by the end of the century, even with the best adaptation efforts. For mid-term fishery management, the results highlight the need for adaptive fisheries management in the light of the ocean warming and acidification expected.

This study has several limitations that can provide new challenges for future research. First, to quantify the effects of ocean acidification this study builds on the experiments reported by Stiasny et al. (2016) that rely on two extreme CO_2 scenarios (i.e., present-day versus end-of-century CO_2). The elevated CO_2 treatment corresponds to a business-as-usual emissions scenario consistent with RCP8.5 (IPCC, 2014). In light of international commitments to mitigate carbon emissions (i.e., to not continue with a business-as-usual approach), it would be useful to explore the implications of more optimistic CO_2 trajectories in terms of impacts on Northeast Arctic cod. Second, because until now experimental work cannot provide data for a case between the two extreme scenarios, it will be important to explore other ways of quantifying the implications of warming and acidification on recruitment. One possibility is to use an ecosystem model, explicitly capturing *in situ* temperature and pH for different depth layers in the Barents Sea. Currently, this study relies on the monthly integrated mean temperature for 0–200 m depth along the Kola Section, as well as general scenarios for atmospheric CO_2 concentrations and so lacks the spatial resolution relevant for cod spawning areas. Third, the modeling approach described here will never be able to capture all real world complexities associated with the socio-economic effects of ocean warming and acidification on an important commercial fishery, such as that for Northeast Arctic cod. On this basis, rather than consider the results obtained here as firm predictions, they should be seen as a possibility of what could happen if unregulated ocean acidification adds to the well-known stressors of warming and overfishing.

Acknowledgments

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Annex 4

The Greenland shrimp (*Pandalus borealis*) fishery

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Contents

A4.1 Introduction

This case study examines the potential impacts of ocean acidification on the Greenland shrimp (*Pandalus borealis*) fishery. It begins by providing an assessment of the biological characteristics of the northern shrimp and considers ways in which ocean acidification may change population dynamics. The annex then goes on to describe the biogeochemical potential for ocean acidification effects in the Baffin Bay / Davis Strait region and the role they might play in the Greenland shrimp fishery. As a case study, the species is contextualized in terms of its socio-economic role in the Greenlandic economy and society. Unusually for living marine species in the region, where subsistence use of resources remains prevalent in many communities, its role is chiefly market-based. This feature, however, simplifies the task of tracing potential societal impacts from ocean acidification or temperature effects initiated perhaps at the planktonic level, and identifying human response opportunities from chemical changes associated with ocean acidification or temperature that act through changes mediated via biological and ecological changes.

The conceptual model in Figure A4.1 illustrates how ocean acidification and other potential climate drivers may affect the Greenland shrimp fishery. Impacts initiating in biogeochemical space interact with ecosystems and economic factors, which overlap. The impacts may stem from external forcing (i.e., the multiple biogeochemical drivers acting through climate change and ocean acidification), but may also come from human activities (i.e., fisheries). For the Greenland shrimp fishery, these drivers include increased ocean acidification (expected to move from north to south), the effects of this increased ocean acidification (expected to move from west to east due to biogeophysical characteristics of Baffin Bay and Davis Strait), and/or temperature and ecosystem changes (expected to move from south to north). They might also include behavioral shifts in the fishing industry, including changes in fishing harvest effort and/or location for shrimp or for ecologically related species, especially cod. These forces combine to determine the production capabilities for ecosystem services and economic rents upon which humans and other ecosystems may depend, and where human behavior can affect resilience. The flow of shrimp and/or other species available for harvest in the Baffin Bay / Davis Strait region may be directly or indirectly affected by these forces. At the same time, the flow of economic rents, or the value derived from the flow of shrimp and related ecosystem services may also be directly or indirectly affected. In particular, changes in the quantity and quality of the shrimp available may be reflected in prices on world markets. Returns to physical capital both onshore and offshore may therefore shift, with broader impacts on the heavily fishing-dependent Greenlandic economy. Investment and related adaptation efforts made in response to these shifts may in turn feed back into the hybrid space and influence resilience.

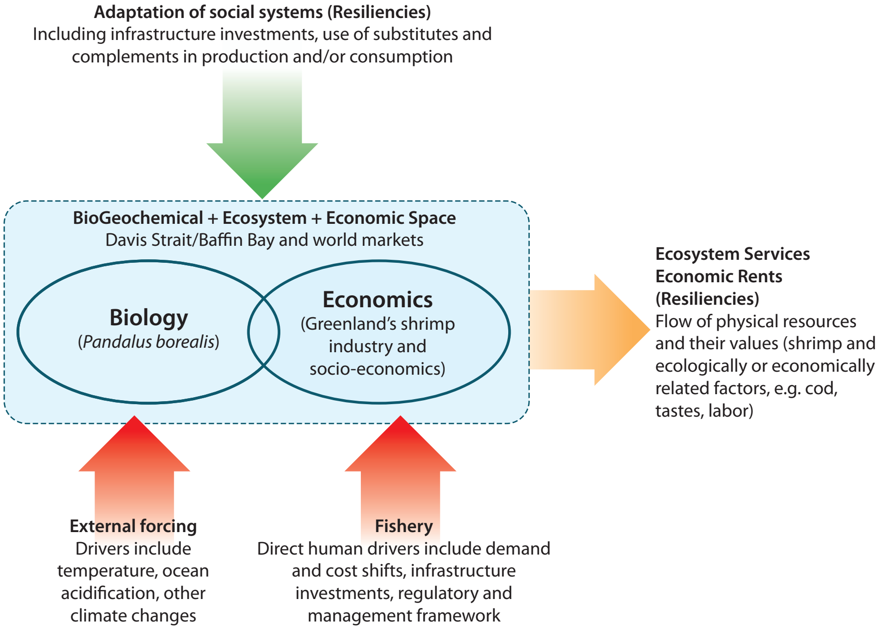

Figure A4.1 Conceptual model of the potential impacts of ocean acidification on the Greenland shrimp fishery and dependent communities.

There are many significant uncertainties in the bio-economic processes of the Greenland shrimp fishery; use of the conceptual model described in this case study helps to summarize how these uncertain human, ecological, and geochemical processes are likely to inter-relate. At present, there are few empirical end-to-end models that are specific to a single fishery, and only a handful of bio-economic analyses that have focused on larger scale or even global assessments of change (Sumaila et al., 2011; Lam et al., 2016). The species- and location-specific work described here should enable scientists and policymakers to identify particular limitations in the current science and management, and should help in decisions about how best to take action in light of the many uncertain impacts and costs of potential ocean acidification and/or related climate factors. The ability to observe the details and interactions between the many uncertainties present at the level of a single fishery serves to shift policy recommendations from narrow and potentially ineffective decisions over harvest or conservation in an individual fishery, to broader actions aimed at both economic and ecological resilience.

A4.2 Biology and potential responses to ocean acidification

Contemporary features

Geographic distribution

Northern shrimp (also known as the pink, deep-water, or deep-sea shrimp; *Pandalus borealis*, Krøyer 1838) is distributed in the colder regions of the North Atlantic and Pacific Ocean (Figure A4.2). This discontinuous circumpolar species occurs on continental shelves in the western North Atlantic from Georges Bank north to Davis Strait, through East Greenland and Iceland to the eastern North Atlantic including the Norwegian Sea, Barents Sea and North Sea. The species has also been reported from the Pacific Ocean near British Columbia, the Gulf of Alaska and Bering Sea, Japan and eastern Russia (Shumway et al., 1985; Bergström, 2000; Garcia, 2007; Wieland and Siegstad, 2012).

Habitat preference

Within the wider geographical distribution, northern shrimp can be found in waters with temperatures ranging from -4°C to 12°C, but is more typically found between 1°C and 6°C. In terms of salinity, these organisms generally occur in areas of between 23 and 36 units (Shumway et al., 1985; Bergström, 2000). While they can live at depths anywhere from 20 to 1400 m, northern shrimp are common at depths of around

Figure A4.2 Distribution of the northern shrimp, *Pandalus borealis* (Bergström, 2000).

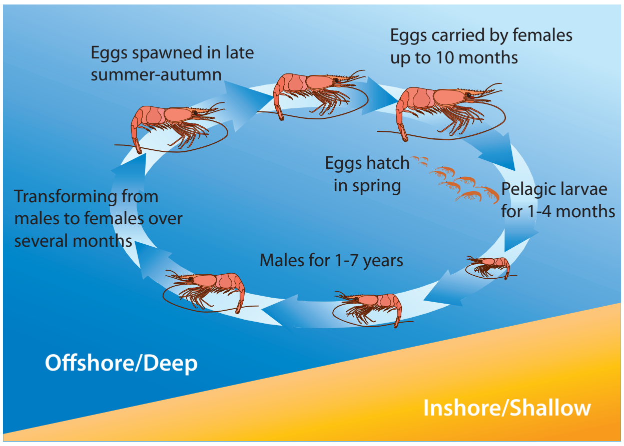

Figure A4.3 Schematic illustration of the *Pandalus borealis* life cycle (Fisheries and Oceans Canada, 2018).

200 to 400 m (Shumway et al., 1985; Bergström, 2000; Dvoretsky and Dvoretsky, 2015). Depth preference does, however, vary throughout the life cycle; shrimp spend their early stages in relatively shallow water (5 to 20 m) often associated with estuaries, with juveniles and adults more common in deeper water between 50 and 500 m (Hammer and Pedersen, 2013). At these depths, northern shrimp are often associated with benthic waters or soft, muddy substrates (Shumway et al., 1985).

Life-cycle history description

Northern shrimp are protandrous hermaphrodites, whose timing of development can vary depending on location (Shumway et al., 1985; Bergström, 2000). Northern shrimp are born male but change sex to female as part of the life cycle – the timing of this transition may affect population dynamics, as significant delays will slow reproduction and vice-versa. In West Greenland specifically, it is typical for juvenile males to mature at about three years of age; transition to female at between five and six years (Wieland, 2004); and then attain an age, as female, of eight or more years, where the maximum remains uncertain (Savard et al., 1994). In reproductively active individuals, mating and spawning occur from July to September. Females brood embryos externally with the egg-bearing period lasting eight to ten months depending on bottom-water temperature. When developed, larvae are released in shallow (<50 m) coastal waters during the following winter (Shumway et al., 1985; Bergström, 2000). Larvae then hatch and go through six pelagic stages lasting between 60 and 90 days in total before settling as benthic juveniles in late spring or early summer (Shumway et al., 1985; Wieland, 2005; Rasmussen and Aschan, 2011). Juveniles remain in coastal waters for a year or more and then migrate offshore to join the mature stock (Shumway et al., 1985). The northern shrimp life cycle is summarized in Figure A4.3.

Interactions with other species

Northern shrimp are generally benthic opportunistic omnivores, but will also feed in the water column during diel (diurnal vertical) migrations (Shumway et al., 1985). Analyses of gut content has identified diatoms, foraminifera, nematodes, polychaetes, crustaceans, mollusks, echinoderms, detritus, and fragments of organisms from other taxonomic groups (Stickney and Perkins, 1981; Bergström, 2000).

In several areas, including the North Atlantic and waters off West Greenland, northern shrimp serve as important prey for various fish species including Atlantic cod (*Gadus morhua*), Greenland halibut (*Reinhardtius hippoglossoides*) and redfish (*Sebastes* spp.) (Parsons, 2005; Savenkoff et al., 2006). Other shrimp predators include marine birds, invertebrates and mammals (Bergström, 2000; Parsons, 2005). Currently, the distribution of northern shrimp and Atlantic cod off West Greenland have little overlap, due to the limited presence of cod (Burmeister and Kingsley, 2015a). The stock assessment model and subsequent management decisions for the West Greenland shrimp fishery do, however, consider the potential for cod as a source of mortality (see Section A4.6 for further discussion of bio-economic implications).

The population dynamics of northern shrimp can also be affected by parasites and pathogens, which include bacteria, trematode parasites, nematodes and crustacean parasites (Bergström, 2000).

Possible direct effects of ocean acidification

Syntheses of responses by marine organisms to ocean acidification suggest many species are vulnerable due to effects on a range of processes (e.g., calcification, thermal tolerance, metabolism, behavior, and reproduction) (Kroeker et al., 2010, 2013; Harvey et al., 2013). However, it is increasingly recognized that there is variation in sensitivity among broad taxonomic groups due to life history characteristics (Kroeker et al., 2013). The crustaceans, including northern shrimp, may be a group able to withstand large changes in ocean acidification conditions, at least in the short term (e.g., Whiteley, 2011; Harvey et al., 2013; Kroeker et al., 2013). Where possible this case study aims to use species- and geographically-specific literature in discussing potential responses of northern shrimp to ocean acidification in the Barents Sea and Baffin Bay region (see Table A4.1 for relevant studies). However, given the current paucity of experimental data this may not always be possible and so studies of similar taxa and regions are used to supplement the discussion.

Physiological processes

A key physiological feature that ocean acidification may affect is acid–base equilibria, or acidosis, of the body fluids. Acidosis can result under ocean acidification because seawater has a greater partial pressure of carbon dioxide ($p\mathrm{CO}_2$), which then diffuses over biological membranes increasing $p\mathrm{CO}_2$ in organisms. Within organisms, CO_2 reacts with body fluids to form carbonic acid which further dissociates and increases the concentration of hydrogen ions (Hammer and Pedersen, 2013). Many crustaceans are typically strong ion regulators, an attribute which is likely to make them less vulnerable to acidosis resulting from increased $p\mathrm{CO}_2$ (Whiteley, 2011; Richards et al., 2015). In terms of northern shrimp specifically, an experimental study has revealed that adults do have a relatively high pH regulatory capacity. Under a time-dependent exposure to hypercapnic seawater (i.e., seawater containing high levels of CO_2; in this case pH 6.86) for up to 16 days the shrimp were able to partially compensate (Hammer and Pedersen, 2013). The shrimp were able to achieve this by accumulating buffering bicarbonate ions (Hammer and Pedersen, 2013), a strategy common in a range of

Table A4.1 Response of northern shrimp to ocean acidification.

Location	Treatment level	Treatment duration	Result of reduced pH	Source
Byfjorden, Norway	pH 8.1, 7.6	34 days	Larval survival not reduced but mean accumulated mortality reduced on day 34, significant delay in zoeal progression (development time)	Bechmann et al., 2011
Hillefjord, Norway	pH 8.1, 7.6	Exposure duration was dependent on the duration of larval development	Increased development time, no significant effect on time to first hatching, hatching success, duration of hatching, feeding rate, or oxygen consumption	Arnberg et al., 2013
Åsenfjord, Norway	pH 6.86	16 days	Accumulated buffering bicarbonate ions, pH regulation achieved without significantly increasing ion-regulating activity of total ATPase and Na^+/K^+-ATPase, oxygen consumption not affected	Hammer and Pedersen, 2013
Gullmarsfjord and Brofjorden, Sweden	pH 8.0, 7.5	21 days	Increased mortality, impaired semi-qualitative measures of appearance and taste but not texture	Dupont et al., 2014

ectotherms countering extracellular acidosis (Whiteley, 2011). Regulation of pH was achieved without significantly increasing the ion-regulating activity of total ATP-ase or Na^+/K^+-ATPase or altering oxygen consumption. These results suggest that northern shrimp has relatively well-developed mechanisms to counteract CO_2-induced acidosis (Hammer and Pedersen, 2013).

Calcification is another process that may be affected by ocean acidification. In crustaceans, however, calcification has been found to be relatively robust under modified CO_2 conditions. While there have not been any studies quantifying calcification in the northern shrimp specifically, other taxa have been studied. Calcification responses to ocean acidification in crustaceans have been diverse, with this process found either to be unaffected, or even to increase (Whiteley, 2011; Harvey et al., 2013). However, it is likely that maintaining or increasing calcification does require additional energy which is being reallocated from another process, such as growth or development (Harvey et al., 2013).

Life-cycle assessment

As detailed earlier, sensitivity to ocean acidification may differ among life history stages, with it being widely assumed that early developmental stages will be the most affected by modified environmental conditions (Arnberg et al., 2013).

Survival of adult shrimp has been reduced under experimental acidification scenarios (Dupont et al., 2014) although it is worth noting that this reduction was observed at high temperatures, where the effects of ocean acidification are expected to be exacerbated. In contrast, adult crustaceans at more moderate temperatures typically show no significant effect of acidification on survival (e.g., those crustaceans considered by Kroeker et al., 2013).

Larval northern shrimp have been unaffected in terms of survival under modified CO_2 conditions relevant for predicted ocean acidification scenarios (pH 7.6) (Bechmann et al., 2011; Arnberg et al., 2013), although it is worth noting that mean accumulated mortality at a particular time point was reduced (Bechmann et al., 2011). Similarly, reduced pH did not result in significant differences in time to first embryo hatching, hatching success, duration of hatching, feeding rate, or oxygen consumption (Arnberg et al., 2013). This species has, however, shown delayed development (Bechmann et al., 2011; Arnberg et al., 2013), which may be attributable to a modified ability to calcify or undertake other physiological processes, such as the maintenance of intracellular pH (Bechmann et al., 2011). Delayed development may modify the amount of time spent in specific environments, affecting interactions with other species (either food or predators, discussed in more detail in the Section *Biotically-mediated indirect effects*).

Human response to modified condition

Changes in northern shrimp to which human populations may respond are also important. In this context, northern shrimp were exposed to two treatments (pH 8.0 or 7.5) for three weeks and then semi-qualitatively scored by a panel of tasters. Exposure to pH significantly reduced the score for appearance and taste, but not for texture (Dupont et al., 2014). As a result, shrimp under present-day conditions had a 3.4 times increased probability of being scored as the best, while shrimp from the acidified treatment had a 2.6 times greater chance of being scored as the worst (Dupont et al., 2014).

Other factors determining sensitivity

Biotically-mediated indirect effects

Shrimp populations are influenced by bottom-up and top-down biotic processes. In terms of bottom-up interactions, the reproductive cycle of shrimp is currently timed such that, under average conditions, larval hatch matches that of the spring phytoplankton bloom so that there are sufficient resources for growth (Fuentes-Yaco et al., 2007). Any alteration in the timing of either component in the future, (i.e., shrimp larval hatch or spring phytoplankton bloom) may modify the availability of food. Food supply may become increasingly important as organisms can potentially avoid the adverse effects of ocean acidification on calcification, growth and condition if they have sufficient resources to sustain the energetic costs of tolerance (Koeller et al., 2009; Richards et al., 2015).

Top-down linkages between shrimp and their predators (such as Greenland halibut and Atlantic cod) have been reported for various parts of the North Atlantic. The anticipated slower development of shrimp under acidification will presumably increase the time shrimp spend in the plankton and thus increase their chances of being predated upon. Consequently, a small delay in development may potentially contribute greatly to an increase in mortality rates of planktonic larvae

(Dupont et al., 2010). The mortality of early life stages of shrimp is already high (Storm and Pedersen, 2003) and additional mortality as an indirect consequence of ocean acidification could affect recruitment to local populations (Arnberg et al., 2013). Moreover, if predator abundance were to be modified under acidification scenarios the altered rate of consumption could have magnified consequences for the abundance of prey species. Thus, direct effects on northern shrimp may be outweighed by indirect effects of changes in the relative abundance of their prey or predators.

Co-factors of interest

Human activities are driving ocean acidification in combination with other environmental changes. In the context of northern shrimp, another key environmental condition being modified is temperature. Organisms specialize within certain temperature ranges and are sensitive to extremes (Magozzi and Calosi, 2015). As development, growth and metabolism are all largely temperature dependent, global warming is also likely to affect such functions (Pörtner, 2001; Koeller et al., 2009). It can therefore be predicted that exposure to temperatures at the limit of thermal tolerance will enhance sensitivity to stress, including ocean acidification (Pörtner and Farrell, 2008; Pörtner, 2010). In the North Atlantic, warming appears to be unfavorable for northern shrimp with increased temperatures during the pelagic larval stage associated with lower recruitment (Richards et al., 2012).

These responses may be complicated where increased temperature is considered in combination with ocean acidification. In experimental manipulations considering the response of northern shrimp to the combination of increased temperature and ocean acidification, the negative effects on development driven by acidification in isolation were absent (Bechmann et al., 2011; Arnberg et al., 2013). That is, in contrast to the predictions outlined above, increased temperature may have a positive effect on shrimp by countering the negative effect driven by CO_2. It is important to note, however, that increased temperature also negatively affected a range of parameters under both current and forecasted CO_2 (e.g., larval hatching time, development, and survival, and metabolic and feeding rates) (Arnberg et al., 2013). As a result, it will be important to consider the combined effect of modified conditions on the specific process of interest.

A4.3 Oceanography of the Baffin Bay and Davis Strait region

Baffin Bay is a semi-enclosed sea bordered by Greenland to the east, Baffin Island to the west, the Canadian Arctic Archipelago to the north and the Labrador Sea to the south (Figure A4.4). The Canadian Arctic Archipelago connects Baffin Bay with the high Arctic Ocean through shallow sills (220 m at Nares Strait), while Davis Strait connects Baffin Bay with the Labrador Sea

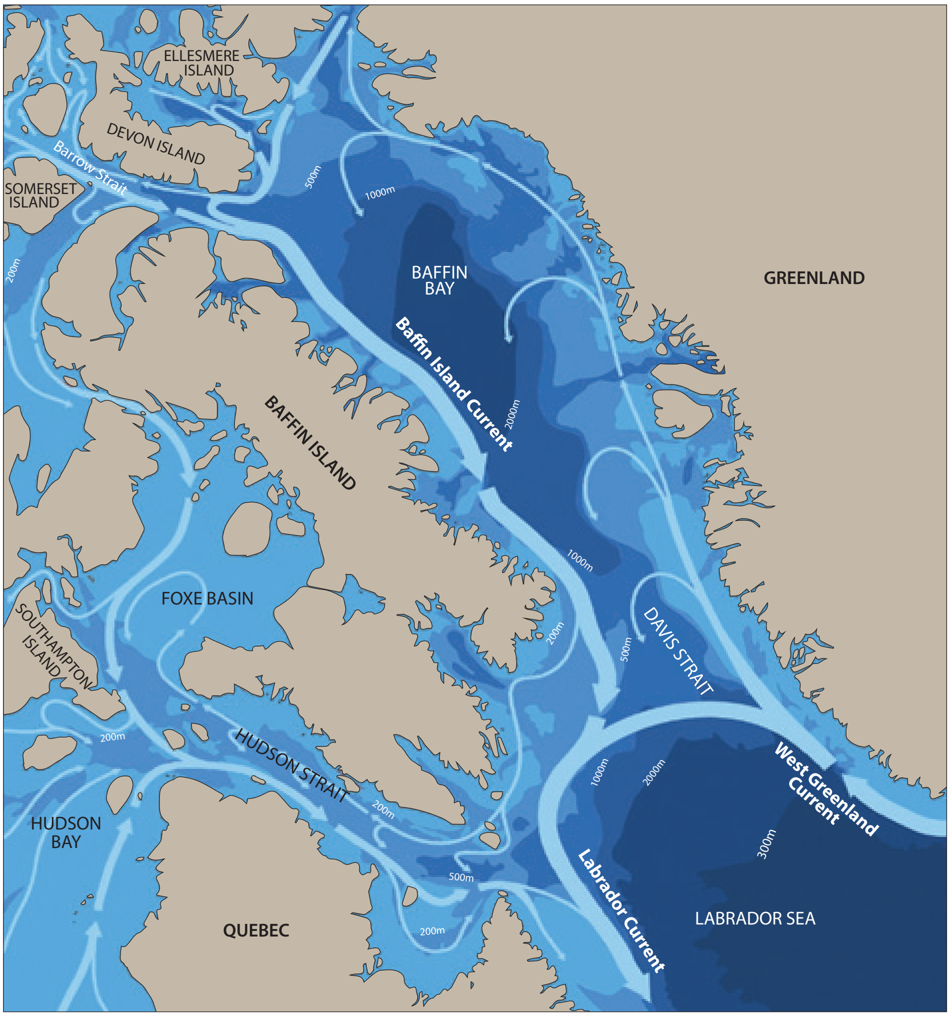

Figure A4.4 Ocean current circulation in the Baffin Bay / Davis Strait region.

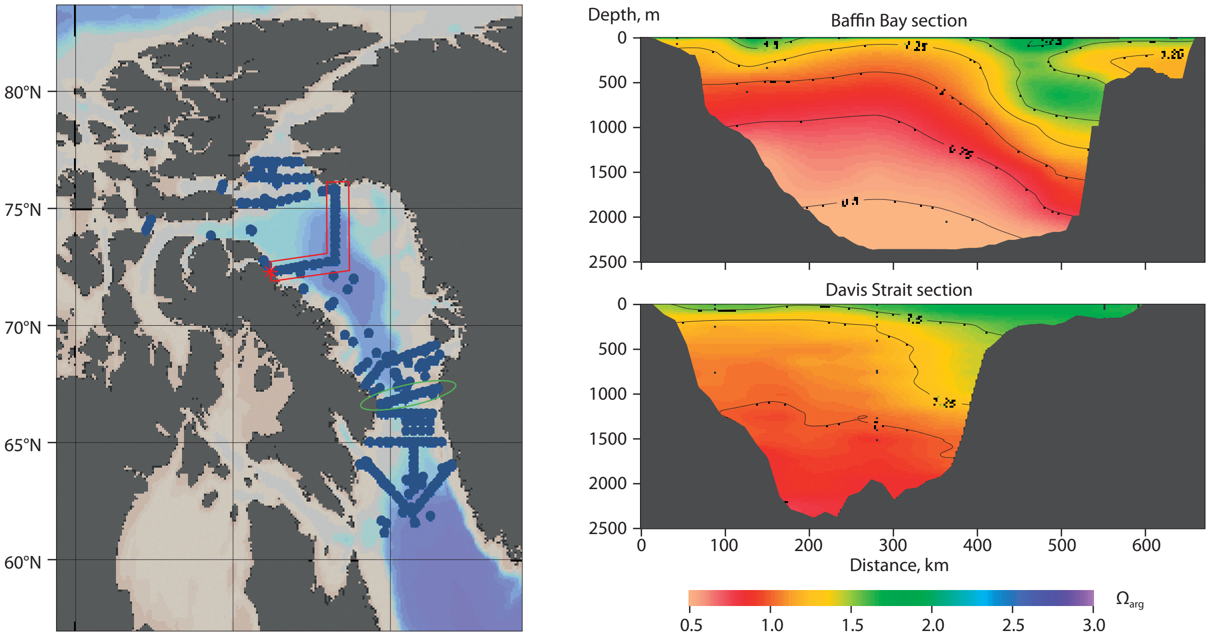

Figure A4.5 Summary of aragonite saturation values in Baffin Bay and Davis Strait for the period 1997–2015. Data are from GLODAPv2 (Olsen et al., 2016) and Azetsu-Scott and Meire (pers. comm.).

through a deeper sill (640 m). Baffin Bay covers an area of 689 000 km^2 with the maximum depth of 2400 m in the central part, and is extensively ice-covered from November to July, except for the North Water Polynya (Pikialasorsuaq) which is a prominent open water and thin ice region of high biological productivity located at Nares Strait.

The main circulation in Baffin Bay is anti-clockwise (Figure A4.4). Two major components are the southward Baffin Island Current and the northward West Greenland Current. The cold and fresh Baffin Island Current is an integrated Arctic outflow through the Canadian Arctic Archipelago, mainly from Nares Strait, Jones Sound and Lancaster Sound. The Baffin Island Current is less saline and hence less dense than the interior waters of Baffin Bay. As a consequence, the current flows southward along Baffin Island, being confined on the continental shelf and upper slope. The current is modified by Hudson Strait outflow and becomes a component of the Labrador Current which continues flowing southward. The West Greenland Current has two branches, a less saline branch (the West Greenland Shelf Current) and a more saline component from the Irminger Sea (West Greenland Irminger water) concentrated on the continental slope. The West Greenland Shelf Current has an Arctic origin and enters the North Atlantic through Fram Strait as the East Greenland Current. This flows south along the eastern continental shelf and slope of Greenland and undergoes considerable modification when it reaches Cape Farewell at the southern tip of Greenland (Sutherland et al., 2009). The East Greenland Current mixes with warm, salty Atlantic waters to form the West Greenland Shelf Current which flows north along the west Greenland Shelf. At Davis Strait, the West Greenland Current can divide, with part of the flow entering Baffin Bay on the eastern side of Davis Strait and contributing to the anti-clockwise circulation in the Bay and part continuing westward as the Labrador Sea anti-clockwise circulation (Tang et al., 2004; Curry et al., 2011; Münchow et al., 2015). The Baffin Island Current and the West Greenland Current are modified in transit by local inflows, ice formation and melting, and mixing with offshore waters. At the central basin between 1200 and 1800 m deep lies Baffin Bay Deep Water below which is Baffin Bay Bottom Water (>1800 m). The origin of these water masses is still debated (Tang et al., 2004).

Ocean acidification in this region was first reported by Chierici and Fransson (2009). Their study showed that in water collected at 8 m depth using a continuous underway system, there was no calcium carbonate undersaturation (i.e., water corrosive to organisms with shells and skeletons made from calcium carbonate) from the southern tip of Greenland to Barrow Strait. Azetsu-Scott et al. (2010) studied several hydrographic sections in the Canadian Arctic Archipelago and the Labrador Sea and found the aragonite saturation horizon ($\Omega_{arg} = 1$), where water becomes corrosive to organisms with shells and skeletons consisting of aragonite (one of the calcium carbonate minerals) located at depths of 500–1500 m in Baffin Bay and 500–700 m in Davis Strait with an east-west gradient. The aragonite saturation horizon is shallower on the Baffin Island side and deepens towards central Baffin Bay and Greenland (Figure A4.5).

This spatial variability can be attributed to the Arctic outflow in the western basin. The Arctic outflow through the Canadian Arctic Archipelago has a high content of Pacific waters, which have a low saturation state. This Pacific water is further modified by sea ice melt/formation and river input, as well as biological activities, consequently lowering the saturation states for aragonite and calcite in the Arctic, and flows through the Canadian Arctic Archipelago to Baffin Bay and Davis Strait (Azetsu-Scott et al., 2010). This decrease in Ω was also demonstrated along a transect from the North Pacific to the North Atlantic (Yamamoto-Kawai et al., 2013).

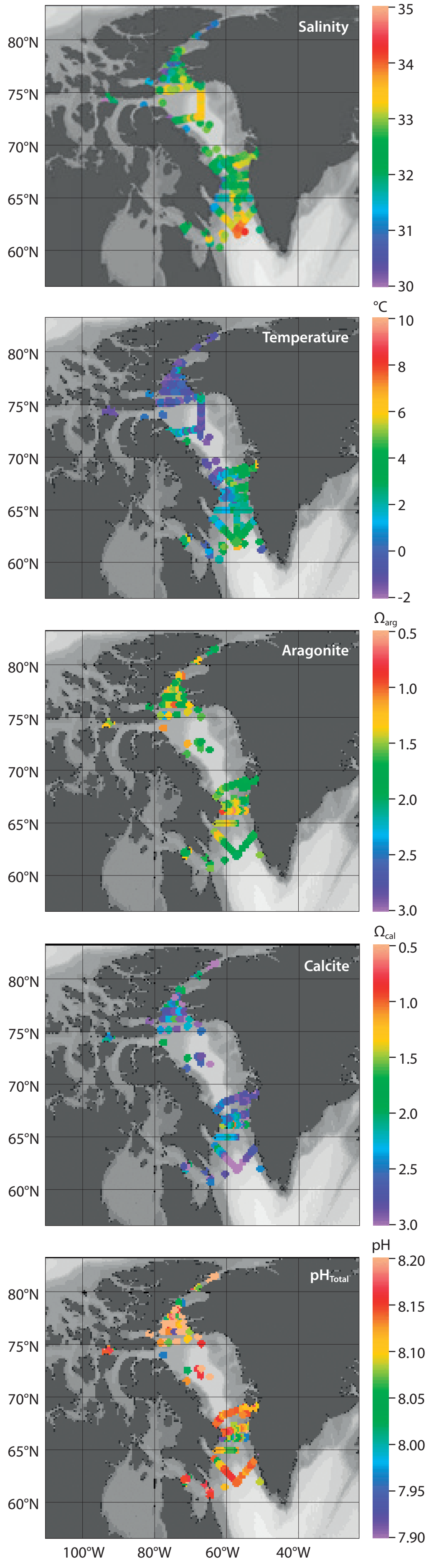

Figure A4.6 Summary of surface water conditions in the Baffin Bay / Davis Strait region for the period 1997–2015.

The low saturation states in the deep Baffin Bay are caused by the limited exchange of deep water in this enclosed basin (Azetsu-Scott et al., 2010). Sills at the north and south limit the exchange of water with that outside the basin. As a result, this deeper water has a long ventilation age (Top et al., 1980) that has led to high dissolved inorganic carbon (DIC) concentrations through CO_2 respiration (Azetsu-Scott et al., 2010).

Summary values of salinity, temperature, calcium carbonate saturation states for aragonite and calcite and pH in surface waters are shown in Figure A4.6. These data were collected during the period 1997–2015 (GLODAPv2 by Olsen et al., 2016; Azetsu-Scott and Meire, pers. comm.). The graphic shows waters to be colder and less saline, and with lower calcium carbonate saturation states in the northern basin than in the Labrador Sea. Arctic water flows through the Canadian Arctic Archipelago with lower calcium carbonate saturation states and pH at the western side of the basin.

Downstream evolutions of salinity, temperature, Ω_{arg}, Ω_{cal} and pH in the total scale (pH_{tot}) in Arctic outflow in the western basin are shown in Figure A4.7. Fresh and cold water with low Ω_{arg}, Ω_{cal} and pH_{tot} is found in the surface waters of Nares Strait, and these properties increase southward. The Arctic outflow, at depths of 150–200 m through Lancaster Sound (Barrow Strait) has the distinctive features of aragonite undersaturation ($\Omega_{arg} <1$) and low pH_{tot}. The higher Pacific water content in the Arctic outflow through Lancaster Sound contributes to the observed lower Ω_{arg}, Ω_{cal} and pH_{tot}, compared to the Arctic outflow through Nares Strait and can be traced to the southern Labrador Shelf (Azetsu-Scott et al., 2010). Along the Greenland Shelf, Ω and pH_{tot} decrease as waters travel northward, together with salinity and temperature at all depths (Figure A4.8). Although no undersaturation of calcium carbonate was observed on the Greenland Shelf, the influence of glacial meltwater on Ω_{arg} was demonstrated in other areas (Fransson et al., 2015, 2016). Glacial meltwater enhances local ocean acidification, although the degree of influence may vary regionally due to local geology modifying the alkalinity of land-derived freshwater such as river inputs and glacial meltwater, and thus the extent of ocean acidification in coastal areas (Azetsu-Scott et al., 2014). Since the Greenland ice sheet is melting rapidly (An et al., 2017), glacial meltwater is the dominant freshwater source in the Greenland Shelf/Slope region (Azetsu-Scott et al., 2012) and amplifies ocean acidification in Greenland coastal areas. The role of kelp forests on pH variability was reported in Greenland fjords. It was demonstrated that marine vegetation effectively modifies temporal and spatial variations of pH in coastal areas (Krause-Jensen et al., 2015, 2016).

It is difficult to evaluate temporal variation in ocean acidification in spatially variable areas such as the Baffin Bay and Davis Strait region without observations at fixed stations/sections over decades (Figure A4.9). Although available data are limited, observations along the Baffin Island Shelf/Slope regions near Davis Strait show that Ω_{arg} has decreased in the surface layers and that the aragonite saturation horizon has shoaled slightly. In contrast, Ω_{arg} has been more variable along the Greenland Shelf/Slope regions. A low Ω_{arg} period between 2006 and 2010 corresponds to a period of low salinity and temperature.

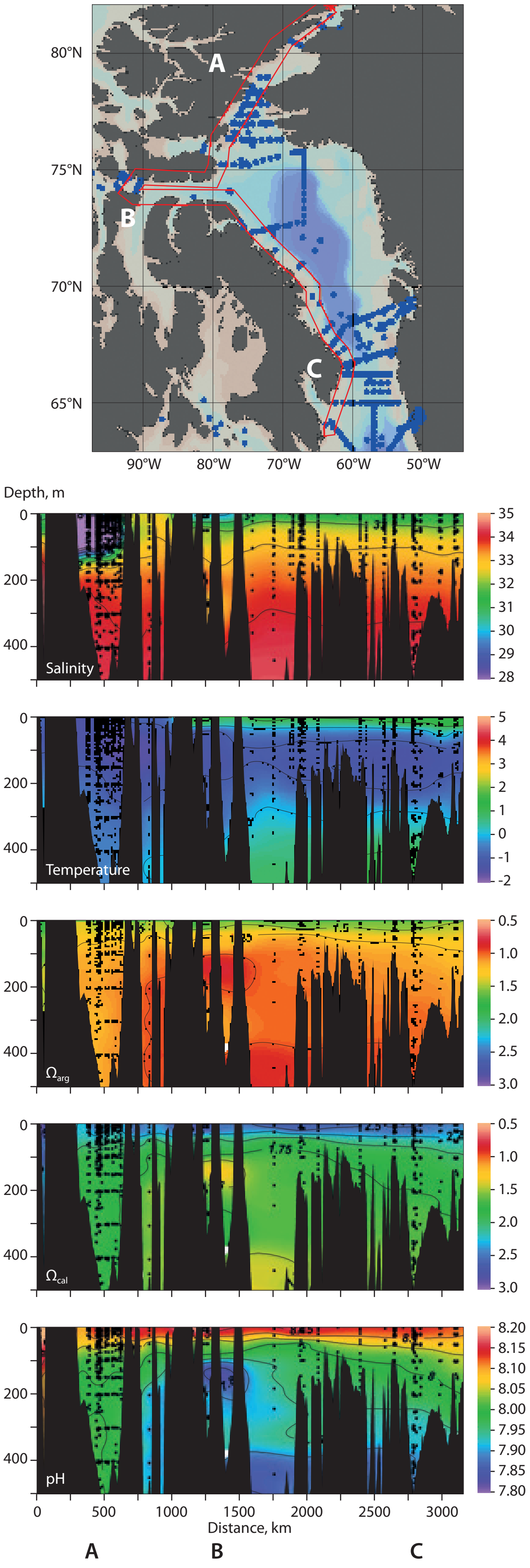

Figure A4.7 Summary images of the Arctic outflow from north to south for the period 1997–2015.

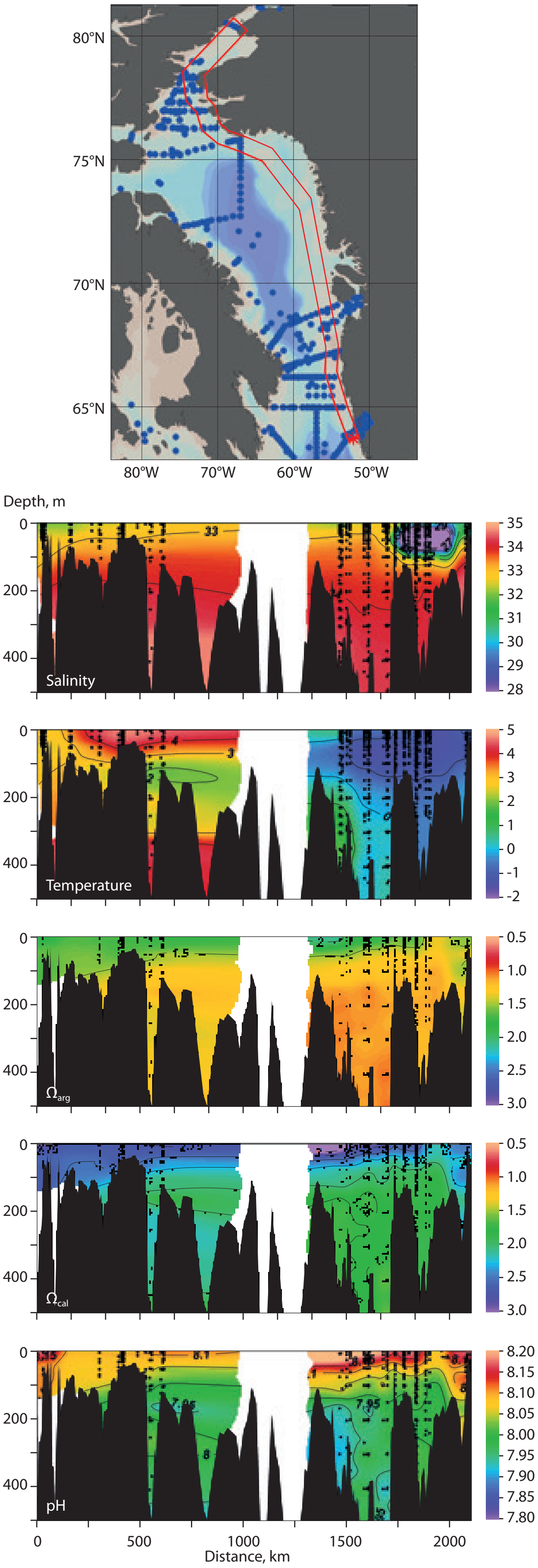

Figure A4.8 Summary images of the Greenland Shelf from south to north for the period 1997–2015.

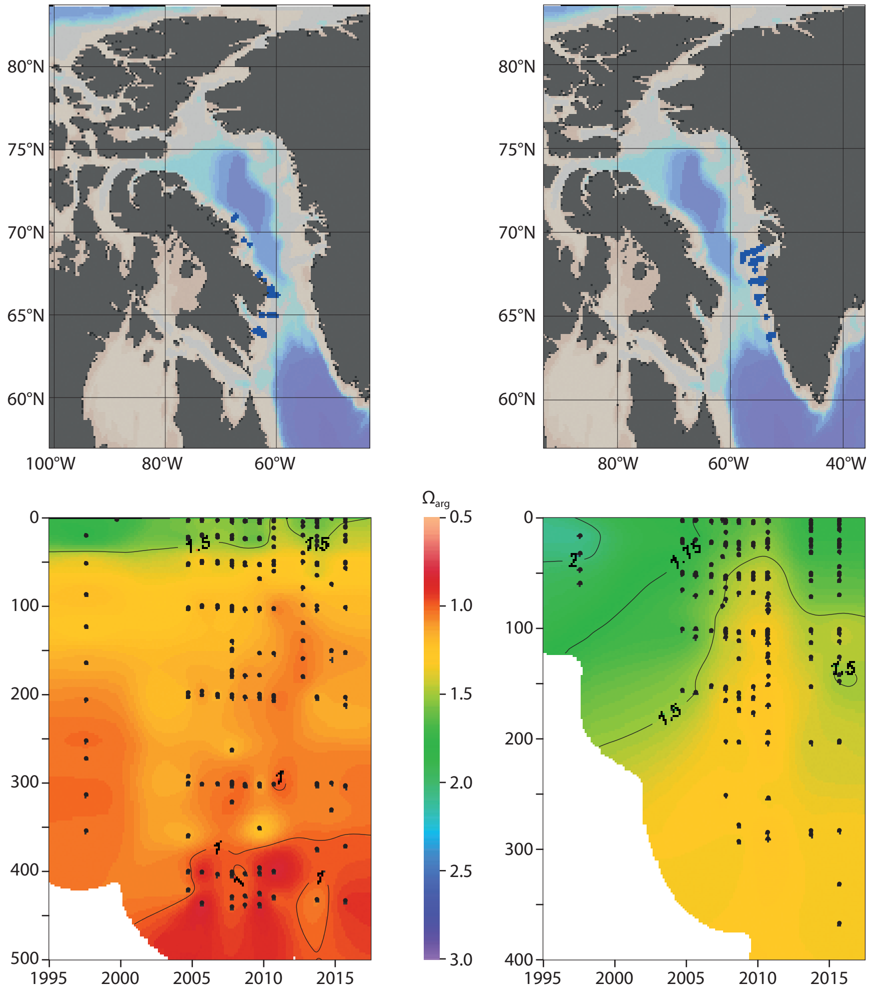

Figure A4.9 Temporal variation in aragonite saturation state (Ω_{arg}) along Baffin Island and Greenland Shelves and Slopes between 1997 and 2015.

A4.4 Biogeochemical linkages

Projected changes in acidification

A multi-model comparison of projection runs using the Intergovernmental Panel on Climate Change (IPCC) RCP8.5 scenario (climate outcome based on ‘business-as-usual’) (Moss et al., 2010) indicates that Ω_{arg} at the ocean surface reaches undersaturation across the entire Arctic region with the exception of the Barents Sea and North Atlantic (Steiner et al., 2014; see Chapter 2, Figure 2.7). The intercomparison includes six Earth System Models (ESMs): the Canadian ESM version 2.0 (CanESM2), Geophysical Fluid Dynamics Laboratory ESM (GFDL-ESM2M), Met Office Hadley Center ESM (HadGEM2-ES), Institut Pierre Simon Laplace low-resolution ESM (IPSL-CM5A-LR), the Japanese Agency for Marine-Earth Science and Technology ESM (MIROC-ESM), and the Max Planck Institute for Meteorology low-resolution ESM (MPI-ESM-LR) (for details see Steiner et al., 2014, their Table 1). A regional analysis shows the simulated pH in Baffin Bay to vary between 8.05 and 8.15 among models for the period 1986–2005 and to decrease to between 7.7 and 7.8 (and 7.9 for one of the models) for the period of 2066–2085 (AMAP, 2013; Steiner et al., 2015). The multi-model ensemble mean trends are -0.043 units/decade (surface), -0.038 units/decade (50–100 m layer), and -0.034 units/decade (100–400 m layer) over a 50-year projection. Inter-model standard deviations are low (between 0.003 and 0.007 units). Future pH would thus decrease by an additional 0.06–0.22 units in the next 50 years. In Baffin Bay, the CanESM2 trend for the aragonite saturation horizon is that it shoals by 60 m/decade for RCP8.5, bringing the mean aragonite saturation horizon close to the surface 50 years from now (Steiner et al., 2015). The models also suggest that more acidified waters will arrive through the Canadian Arctic Archipelago and enter Baffin Bay (Figure A4.10).

A simulation with a higher resolution model (NAA-CMOC) allows for more detail in the horizontal structure and indicates clear differences between western and eastern Baffin Bay, suggesting the outflowing more-acidified waters to stay on the west side of the Bay. NAA-CMOC is a regional model for the Arctic and is described by Steiner et al. (2016). It is forced with output from the 22-km resolution Canadian Regional Climate Model version 4 (CanRCM4) (Scinocca et al., 2016) for RCP8.5.

NAA-CMOC represents only one model representation and so is prone to higher uncertainty. While the spatial structure is likely to be more realistic than that generated by ESMs, the absolute values must be considered with caution. The model has been shown to experience a drift towards higher salinities over time (Hu and Myers, 2013) which affects pH and saturation

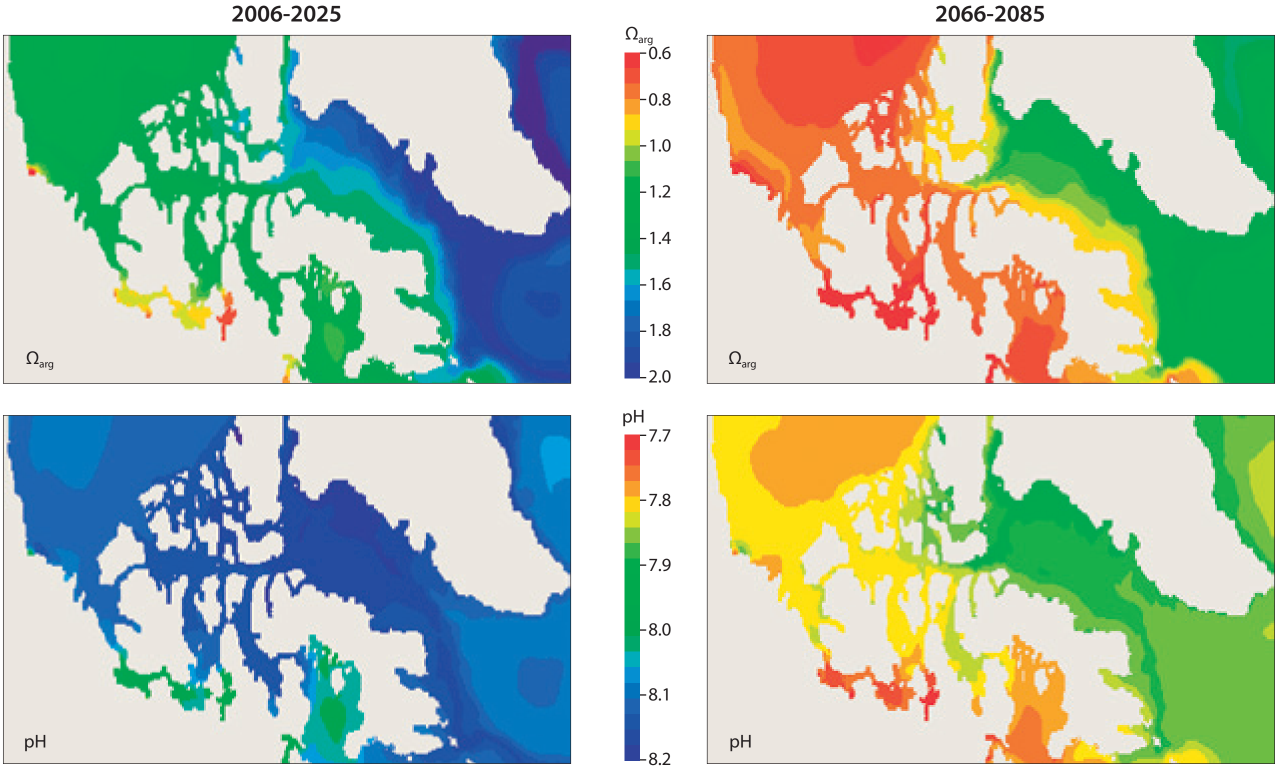

Figure A4.10 Simulated aragonite saturation state and pH in the Canadian Arctic for the periods 2006–2025 and 2066–2085 under the IPCC RCP8.5 scenario.

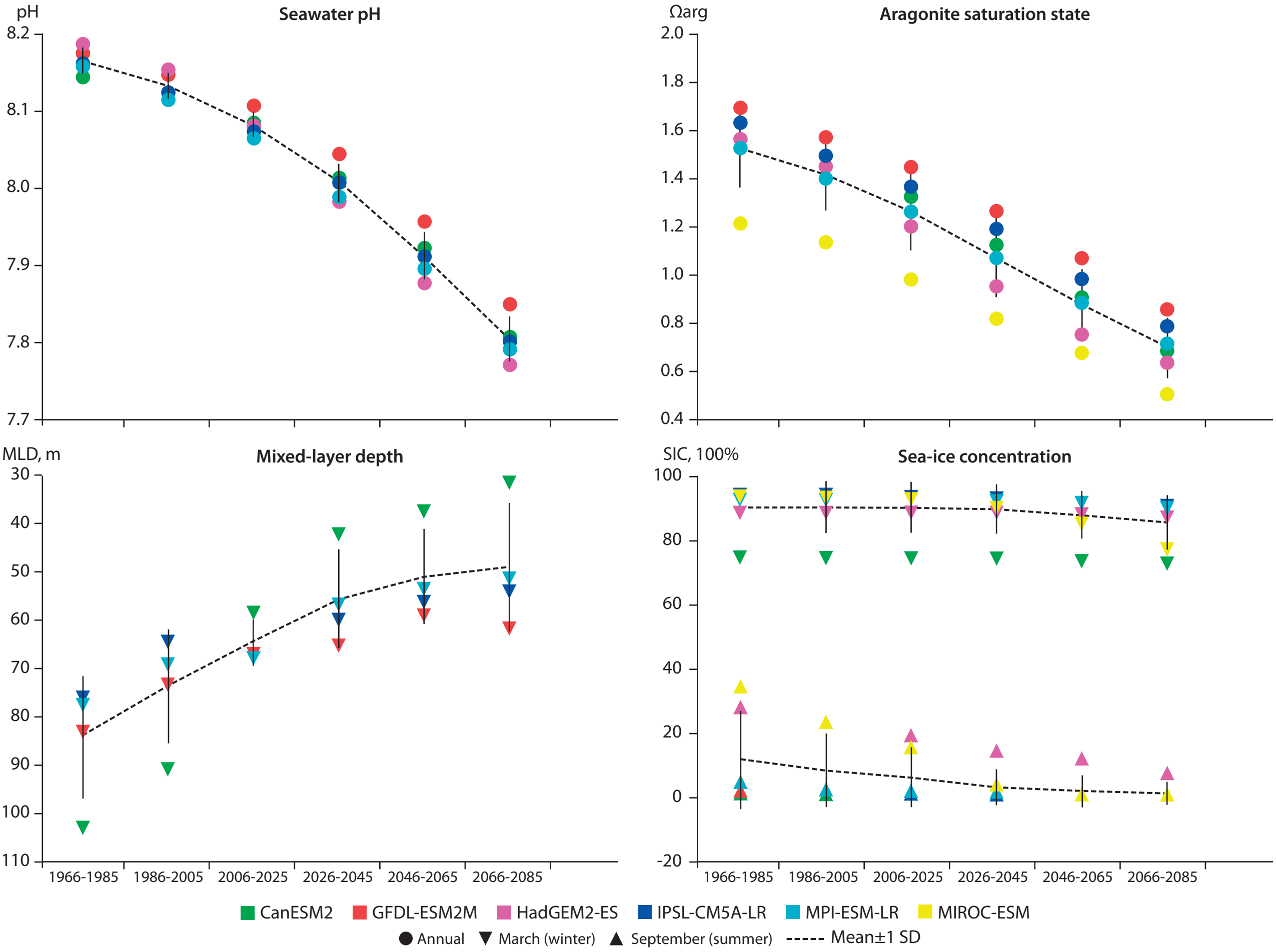

Figure A4.11 Multi-model comparison of bi-decadal averages in annual mean surface ocean pH, annual mean aragonite saturation state, maximum mixed layer depth in March, and sea-ice concentration in March and September under the IPCC RCP8.5 scenario for Baffin Bay (~65–77°N, 60–78°W). The graphic shows individual model averages as well as the multi-model mean and ±1 standard deviation. Note that pH is not available for MIROC-ESM and mixed layer depth is not available for MIROC-ESM and HadGEM2-ES.

state. The driving Canadian ESM, CanESM2, is in the mid-range of the ESMs with respect to projected acidification (Steiner et al., 2014). The future trend has been shown to be very consistent among the models and is likely to be robust and similar between the regional model and the ESMs. Changes from the period 2006–2025 to 2066–2085 are about 0.25 units for pH and 0.6 units for Ω_{arg} (Figures A4.10 and A4.11).

Figure A4.11 shows the multi-model comparison of bi-decadally averaged annual mean surface pH and Ω_{arg}, for the domain of Baffin Bay. Consistency in the projected decrease in pH is apparent. Ω_{arg} shows a corresponding decrease, although inter-model differences are larger in Baffin Bay than in other regions of the Canadian Arctic. One of the key findings from the previous AMAP assessment (AMAP, 2013) is that ocean acidification impacts must be assessed in the context of other changes happening in Arctic waters because ocean acidification is one of several factors (stressors) that may contribute to changes in species composition. Figure A4.11 also shows maximum mixed layer depth in March, and sea-ice concentration in winter (March) and summer (September). Mixed layer depth shows a consistent decrease over time, which indicates an increase in stratification that may be linked to a reduction in nutrient supply to the upper ocean. No major sea-ice decline is visible in Figure A4.11, this is mainly because summer sea-ice cover is generally low in this area, but the ice cover does in fact decrease from about 8% to 1% in Baffin Bay. The winter ice cover also shows some decrease, from 90% to 87% (changes are from 1986–2005 to 2066–2085). The other major factor is an increase in ocean temperature. While a slight cooling trend was observed over the past 50 years in the top 50 m, warming trends have been observed in deeper waters (Zweng and Münchow, 2006; Hamilton and Wu, 2013). Model projections indicate warming of 0.17°C /decade for RCP8.5 by the global ESMs, but about twice that trend for higher resolution models with more warming in southern Baffin Bay compared to the northern part (Steiner et al., 2015).

Projected changes in nutrients and primary production

Multi-model mean changes between 1980–2000 and 2080–2100 for 11 models show a decrease in surface nitrate concentration of 1–2 mmol/m^3 in Baffin Bay and a decrease in specific primary production of up to 6 gC/m^2/y in the northern Baffin Bay and up to 14 gC/m^2/y in the southern Baffin Bay (Figure A4.12, Vancoppenolle et al., 2013). Vancoppenolle et al. (2013) pointed out the large variability in nutrient availability and corresponding primary production in this region. Consistent mechanisms operate for all models, balancing an increase in available light with a decrease in nutrients under a future climate. For Baffin Bay, 80% of models agree on a projected decrease in both nutrients and specific primary production.

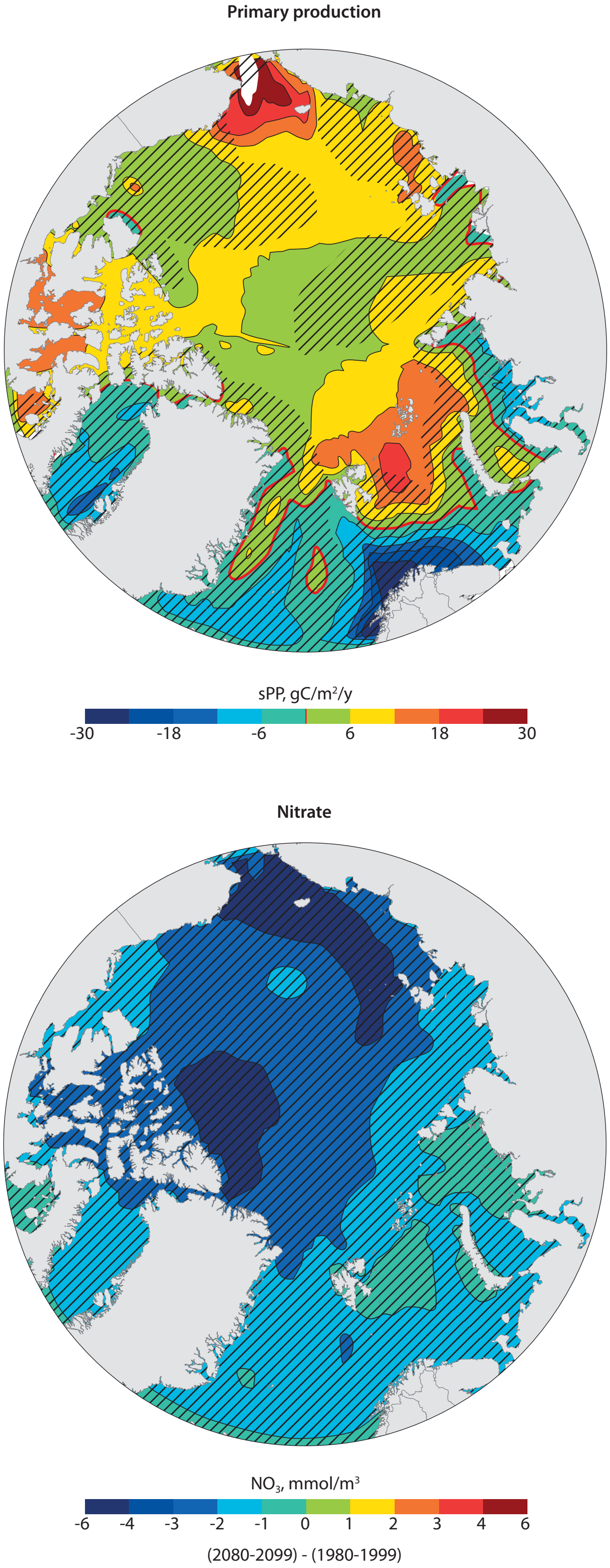

Figure A4.12 Summary of mean model differences between the periods 2080–2100 and 1980–2000 for 11 CMIP5 ESMs. Plots show change in annual specific primary production and surface nitrate concentration. Hatching indicates where 80% of the models agree on the sign of the change (Vancoppenolle et al., 2013).

A more detailed analysis of the Baffin Bay region with a subset of ESMs shows divergent trends in nitrate (NO_3) concentration at the surface and in the 50–100 m layer, and no clear conclusions on projected trends (Steiner et al., 2015). However, limited NO_3 observations in Baffin Bay make it difficult to determine the representativeness of the ESMs in this region (see Lavoie et al., 2013). In Baffin Bay, HadGEM2-ES NO_3 concentrations at depth are too high, while GFDL-ESM2M and MPI-ESM-LR surface concentrations are too high compared to limited observations at the surface.

A4.5 Fisheries conditions and community impacts

Overview of the fishery

Both Canada and Greenland host fishing activities for northern shrimp in Baffin Bay / Davis Strait waters. The quantitative analysis in this case study focuses primarily on the Greenland fishery, while the potential roles of the Canadian fishery in the development of ocean acidification impacts and human responses are discussed from a qualitative perspective. These approaches for the different regions primarily reflect data constraints.

The Northwest Atlantic Fisheries Organization (NAFO) and the International Council for the Exploration of the Sea (ICES) provide the Greenland shrimp fishery with advisory support through annual stock assessments and reports on conditions governing the fishery dynamics. Figure A4.13 shows the NAFO divisions with harbors, graduated by harbor size.

While Canada and Greenland have no formal cooperation over the stock and determine their total allowable catches (TACs) separately, there is some overlap of habitat in Division 0A. Greenland assesses the TAC for Division 0A with this consideration in mind, reserving a proportion of the catch that the Canadians are expected to harvest. This reserved percentage is small, around 3% or less (Arboe, 2014).

There is both an offshore fleet and a coastal fleet fishing northern shrimp in these waters, each facing different restrictions. In effect, the coastal fleet mainly fishes in NAFO Divisions 1A and 1B (north of the 66°N line), while the offshore fleet fishes in all of NAFO Division 1, but not in the inshore areas (within 3 nm of the coast, and in certain reserved 'shrimp boxes', see Arboe 2014, their figure 4). Individual transferable quotas (ITQs) are divided among the offshore Greenland fleet and the coastal fleet, with a proportion of the TAC going to EU offshore interests. (The EU quota generally goes to a single vessel which is treated as part of the Greenland offshore fleet in the analysis; note that any EU vessels are not direct participants in the Greenland ITQ program, rather their share is a separate concession, see NAFO/ICES, 2016). The government, in consultation with scientists from the Greenland Institute for Natural Resources, sets the TAC and then determines individual allocations and apportions these to license holders (Statistics Greenland, 2017a). Licenses

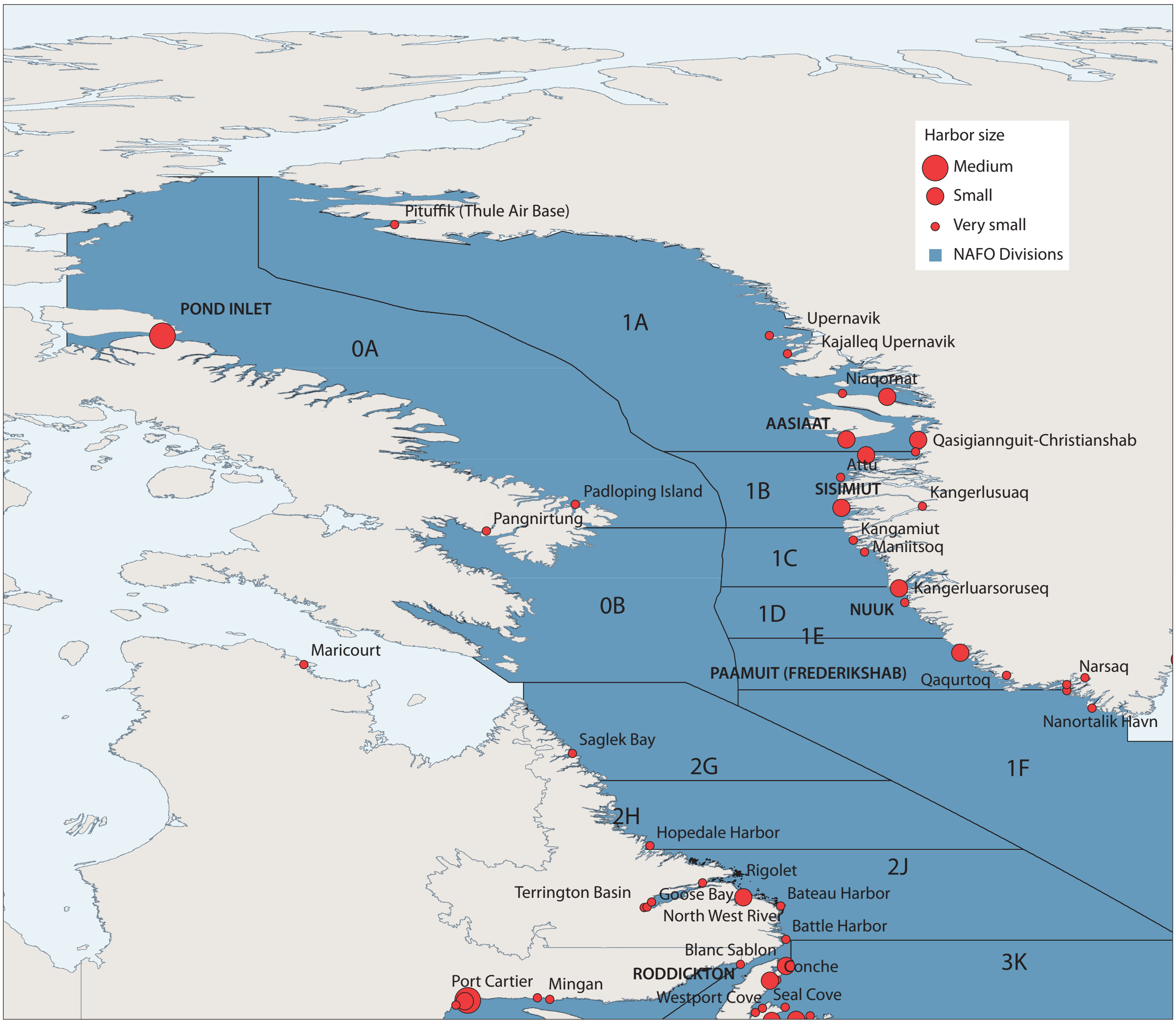

Figure A4.13 Northwest Atlantic Fisheries Organization (NAFO) Divisions in Baffin Bay and Davis Strait, and harbors graduated by size.

Table A4.2 Volume and value of the northern shrimp industry in Greenland, 2013–2016. Data from Statistics Greenland (2017b).

	2013	2014	2015	2016
Total northern shrimp delivered to first sale, tonnes	45,704	42,590	33,788	42,604
Value of total northern shrimp, thousand DKK	351,863	307,770	332,803	399,379
Offshore northern shrimp delivered to first sale, tonnes	13,146	12,882	8,454	7,611
Value of offshore northern shrimp, thousand DKK	68,925	83,720	76,566	82,246
Proportion of Greenlandic seafood delivered, %	44	40	32	34
Proportion of northern shrimp from offshore fleet, %	29	30	25	18

are held in perpetuity and assigned annual quotas; licenses are tradeable (Berthelsen, 2014; Statistics Greenland, 2017a). The coastal fleet processes their catch onshore, while at least 25% of the offshore Greenland catch must be processed onshore in Greenland. Concentration in the coastal fleet ITQs is limited so that no entity can own more than 15% of the near-coast quota.

Economic importance of the fishery

Overall, northern shrimp has accounted for about 32–44% of fisheries' value in Greenland for the past five years (Statistics Greenland, 2017b). Fisheries have historically comprised about 85% of Greenland's exports, accounting for about 55% of the country's market economy and 20% of Gross National Income (Vestergaard et al., 2010). Since 2013, in the Greenland offshore fishery, both the total volume of northern shrimp and its share have decreased, although their value on the market and the overall catch has remained relatively steady or increased (Table A4.2).

Total catches since 1975 are shown in Figure A4.14. Catch has fluctuated over the years, with an annual mean of ~92,500 t (SD ~34,250 t). The current reductions in catch reflect increased management (Arboe, 2014).

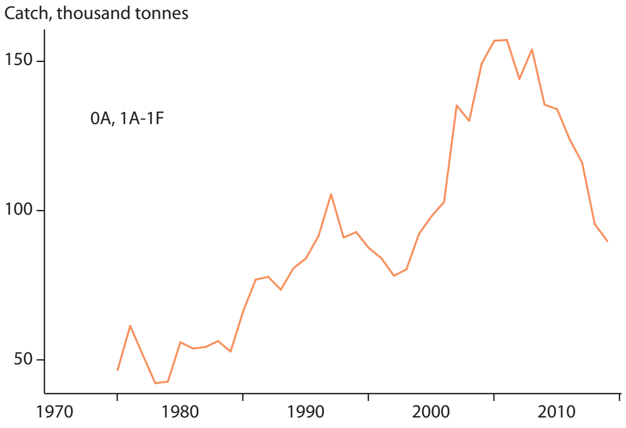

Figure A4.14 Total annual catch of northern shrimp in NAFO Divisions 0A, 1A-1F. Data from Arboe (2014).

Changes in fishery productivity and location

Catch per unit of effort (CPUE) across the seven NAFO divisions in West Greenland is illustrated in Figure A4.15, and catch, effort and CPUE for each of these seven regions are compared in Figure A4.16. CPUE is a measure of the effectiveness of the fishery and may vary over time due to changes in technology (governing how easy it is to catch the shrimp) or to the availability of shrimp in the area. Catchability has indeed been increasing in the fishery and in recent years larger vessels (>80 m) have been allowed to fish in coastal waters and the offshore fleet has increased its use of double trawls (Arboe, 2014; NAFO/ICES, 2016).

Looking at the catch, effort and CPUE data together (Figure A4.16) it is apparent that the different NAFO Divisions have experienced different impacts over recent decades. The CPUE used here is the unstandardized CPUE (i.e., agreed kg/h). In particular, shrimp fishing has become less prevalent in the southern Divisions (1C–1F) and Division 0A, while CPUE in these southern Division 1 areas is high and has increased. As illustrated in Figure A4.15, however, CPUEs have dispersed over the years and the regions still being regularly fished (1A and 1B,

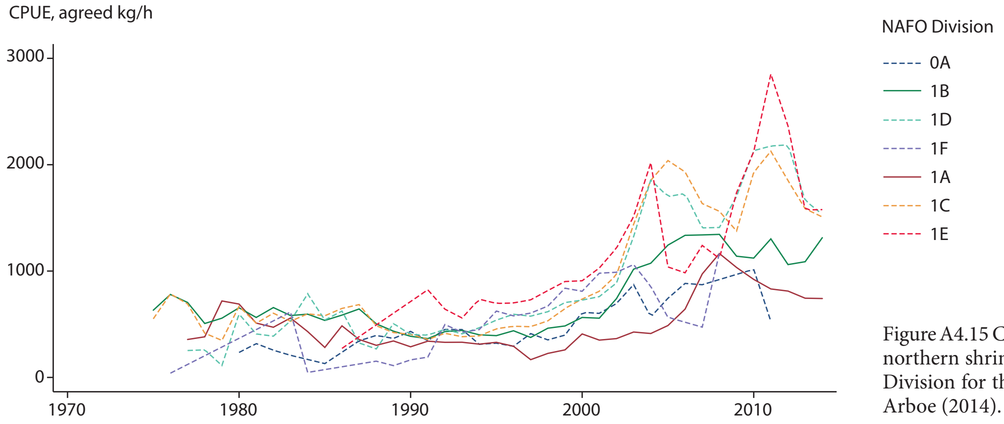

Figure A4.15 Catch per unit of effort (CPUE) for northern shrimp in West Greenland by NAFO Division for the period 1975–2014. Data from Arboe (2014).

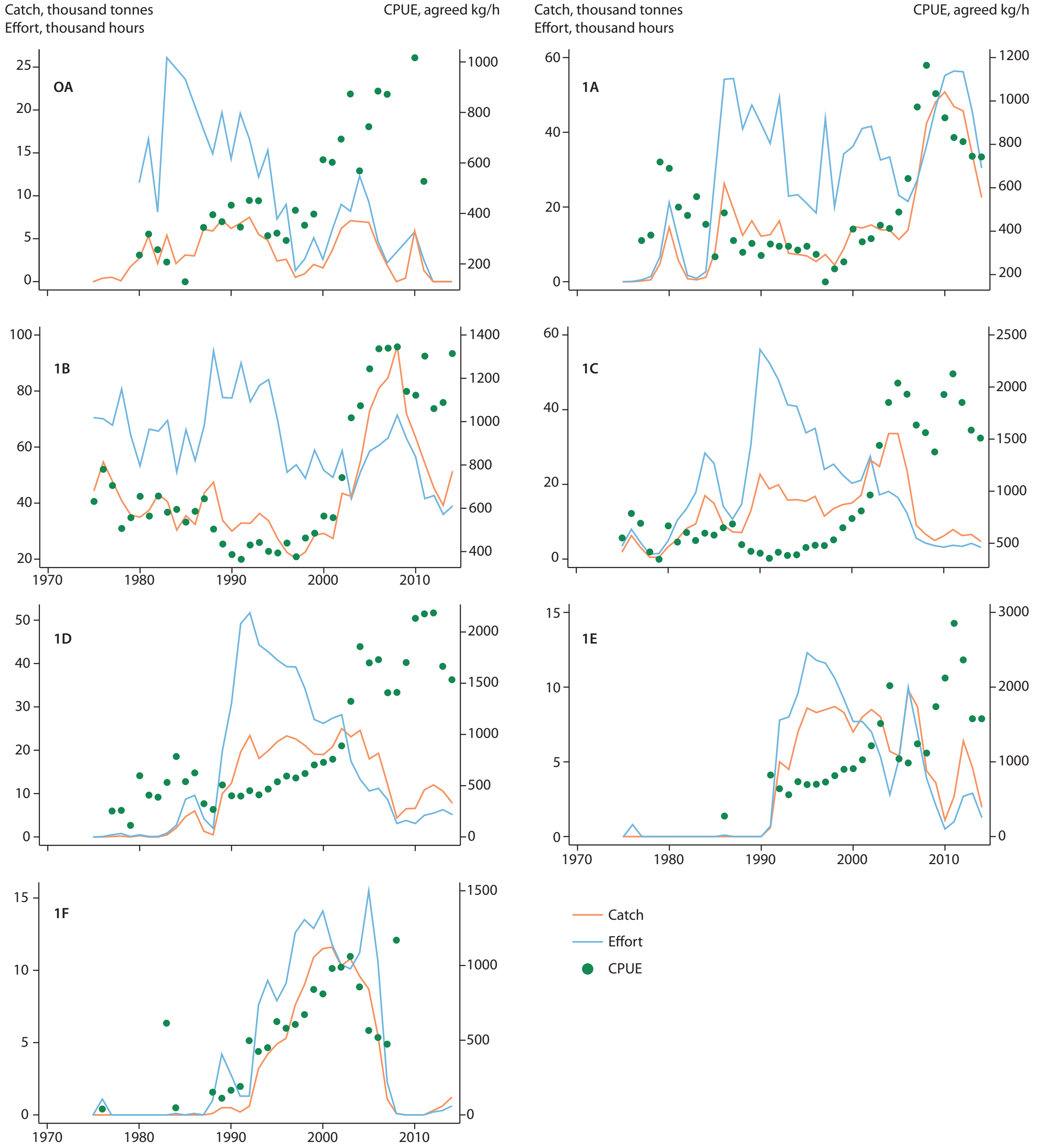

Figure A4.16 Comparison of catch, effort and catch per unit of effort (CPUE) for northern shrimp in West Greenland by NAFO Division for the period 1970–2014. Data from Arboe (2014).

shown with solid lines) now have lower CPUEs than any other area besides Division 0A. This suggests that the constraint is not the availability of shrimp in the region but some other factor. Otherwise, more fishing could be expected in the higher CPUE areas in order to increase yield per hour and therefore profit. While these human behavioral responses are not currently targeting ocean acidification or related co-factors from climate change, they do assist in highlighting where resiliencies and stress-points within the industry are located.

When the data are reported differently (for offshore/coastal fleets rather than by NAFO Division) and include 2014 (actual) through the first part of 2016, they show increased regional convergence in the north beginning in 2014 (NAFO/ICES, 2016: their figure 3.2); this convergence excludes a continued decline essentially to zero for the Canadian fleet (NAFO Division 0A). The convergence to the north reflects a repositioning of fishing harvest resources and effort to improve profits by reducing harvesting costs. This has caused some concern to scientists, however, who are uncertain about its effect due to modelling the stock as a single population (Arboe, 2014). The changes in spatial concentration may mean that the fleet data are not reflecting consistent biological conditions over time and that estimates of fishing mortality intended to reflect the entire population do not adequately accommodate concentrated pockets of fishing pressure (NAFO/ICES, 2016).

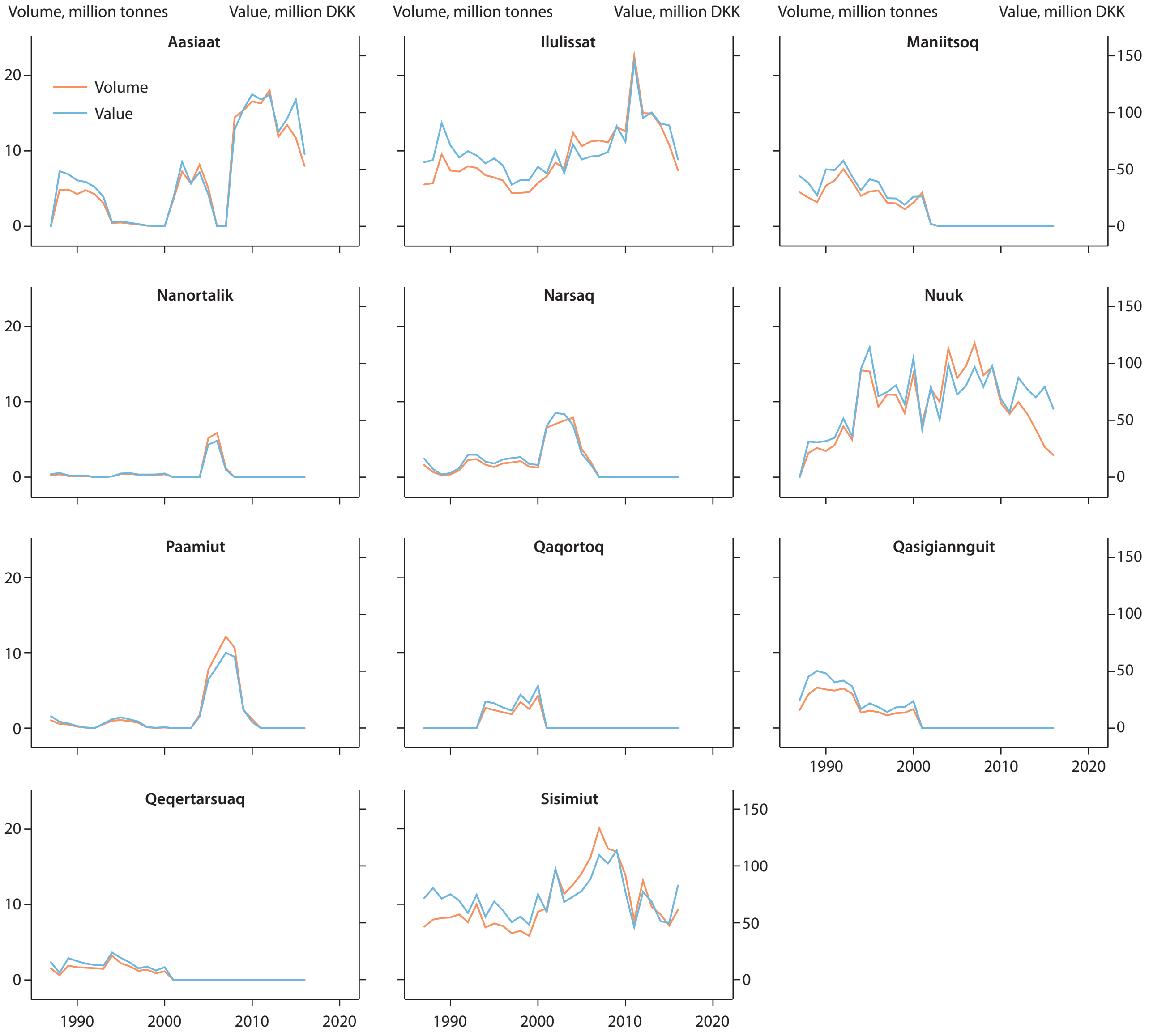

Figure A4.17 Volume and value of onshore Greenland northern shrimp processing for the period 1986–2016.

Community impacts from commercialization

Since the early 2000s, the number of processing locations in West Greenland has reduced from 11 to four. This reduction in the number of facilities, together with the ongoing requirements for onshore processing creates impacts which differ spatially across communities, both for the fishing fleet and for the onshore processing, which generate both jobs and income. Onshore fish and shrimp processing in Greenland has a long history that has formed an integral part of Greenland's market economy; at least two of Royal Greenland's processing facilities have been operating for about a century, with Sisimiut starting in 1914 (most recently improved for shrimp in 2011) and Ilulissat starting in the 1920s (most recently improved for shrimp in 2010). Others, such as Qeqertarsuaq which started in 1934, have switched from shrimp processing to other species (Royal Greenland, 2017).

While government-owned Royal Greenland is the world's largest processor of cold water shrimp and Greenland's largest employer, it is no longer the only company processing shrimp in Greenland. Polar Seafood, the most significant competitor also has plants in Nuuk and Aasiaat. Figure A4.17 shows the volume and value of the inshore catch and processing for the 11 locations actively processing shrimp between 1986 and 2016 (data from Statbank Greenland, 2017). The high variability across locations corresponds to the overall fluctuations evident in Figure A4.14 and only Aasiaat (NAFO Division 1A), Ilulissat (NAFO Division 1A/1B), Nuuk (NAFO Division 1C/D) and Sisimiut (NAFO Division 1B) remain active today. The overall declines since 2012 also reflect decreasing quota levels over this period; there is less quota being shared among fewer locations.

The data described here regarding the increasing spatial concentration of the shrimp fishing activities (Figure A4.16) suggest that the decline in the number of locations processing shrimp is more a function of the commercialization of the industry than of the location of the shrimp, although there has been some overall reduction in catch since the mid-2000s (Figure A4.14). Several points may be taken from this: (1) economic factors have significantly changed the fishery over the past century, and can also be expected to matter in the future; (2) the distribution of impacts may be driven by ecological or economic factors; (3) these impacts and their distribution reflect significant responsiveness of the fishing industry but also volatility and losses for the onshore communities where processing has reduced or disappeared; and (4) understanding the long-term impacts of ocean acidification and/or other climate factors on the northern shrimp industry in Greenland, or any living marine resource requires a simultaneous consideration of economic and ecological factors.

A4.6 Ocean acidification and the socio-economics of the Greenland shrimp fishery

Overview of bio-economic modelling

Integrating the biological, ecological and geochemical elements of the natural world with the economic elements of the human world (as shown in Figure A4.1) requires model building that captures potential changes in both sets of systems in ways that elucidate potential impacts and their risks. Sections A4.2, A4.3, and A4.4 established the initial biological and oceanographic conditions and the most likely changes that may influence the northern shrimp fishery. Section A4.5 examined the past and present economic conditions in the fishery that determine the framework through which these impacts may be felt. In this section, both sets of information are used to model the potential impacts and uncertainties.

Theoretical model for the effect of ocean acidification on the Greenland shrimp fishery

Ideally, the potential socio-economic impacts of ocean acidification on the fishery could be modelled in four steps. If they could be completed, these would be:

1. Provide a cohort model of somatic growth, to determine how changes in rate of growth would affect the biomass available.
2. Use sex-specific information from that cohort model to determine how population level changes in the timing of the transition from male to female might evolve from ocean acidification and in turn affect biomass available for harvest.
3. Determine natural (such as from cod predation) and fishing mortalities and use these in combination with '1' and '2' to calibrate a model of the fishery ecosystem biodynamics.
4. Incorporate the impacts of incentives from prices and rents on fishing mortalities to create a fully bio-economic dynamic model of the fishery and the potential for ocean acidification impacts.

In pursuing these steps, the uncertainties inherent in scientific modeling will be compounded. In the case of the Greenland shrimp fishery, the uncertainties are significant. It is already apparent from the discussion in Section A4.2 for example that population effects are not well studied for northern shrimp, so any results will be limited by having been extrapolated from findings regarding other species and locations.

The results of the fully developed bio-economic model are not likely to provide clear-cut recommendations for shifts in the current management of the fishery. Nevertheless, by attempting to calibrate and run such a model, the limitations and gaps that most need additional research are becoming more apparent.

Current stock assessment model of the fishery

Annual stock assessment models for the West Greenland shrimp fishery fall within the purview of the Greenland Institute for Natural Resources. These inform NAFO/ICES (NIPAG) meetings and reports on *Pandalus* stocks more generally.[1] The projections of the fishery stocks in the coming years are given as advice to the state government, which uses them to decide on the TACs for the NAFO Divisions. The primary data for the models are from logbooks of the fleets. In addition to the fleet data, the stock assessments are informed by an annual Greenland trawl survey that has covered the offshore fishing grounds since 1988 and the coastal areas since 1991 (NAFO/ICES, 2016).

The stock assessment model is a quantitative surplus production (Schaefer) dynamic model based on 30 years of data on the West Greenland stock of *P. borealis* to forecast future stock levels (Kingsley, 2015), and has been in use, with improvements, since its development in 2002 (Hvingel and Kingsley, 2006). The Schaefer surplus production model originated in the mid-1950s. It is not age-structured and has been attractive due to its simplicity and applicability. Prager (1994) reviewed the modelling and exposition of limits and opportunities for remediation.

These stock assessments are state-of-the-art in fisheries management, in that they are annual, combine survey and fleet data, and include consideration of multi-species ecosystem impacts. Of over 1200 marine fish stocks that are managed globally only 24 have been identified as formally including ecosystem approaches in the stock assessments; and northern shrimp is one of the 24 (Skern-Mauritzen et al., 2016). The model includes a cod predation factor, so that mortality in the fishery includes cod predation and fishing mortality. Northern shrimp and cod do not always overlap and the relationship off West Greenland is not constant. Because the model includes cod interactions it integrates data on Atlantic cod from other sources, including the German groundfish surveys (NAFO/ICES, 2016), such that the expected effective cod biomass can be used to determine the expected mortality factor for northern shrimp.

There has been large variability both in the amount of cod overlap with the shrimp and with the survey estimates of the total amount of cod. In 2016, for example, estimates of effective

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[1] NAFO and ICES also prepare assessments for the other main northern shrimp fisheries in the North Atlantic. These are on the Flemish Cap (NAFO Div 3M), on the Grand Bank (NAFO Div. 3LNO), in Denmark Strait and off East Greenland (ICES Divs XIVb and Va), Skagerrak and Norwegian Deep (ICES Divs IIIa and IVa east), in the Barents Sea (ICES sub-areas I and II), and in the Fladen Ground (ICES Div IVa).

overlapping cod biomass are as little as 5% of the 2015 estimates (i.e., 3 vs 60 kt) (Burmeister and Kingsley, 2016b; NAFO/ICES, 2016; Siegstad, 2016). The assessment treats this variability with caution; rather than assuming there will be few cod eating few shrimp so that shrimp quotas might be high, an average predation of recent years of 35 kt is used. This gives a more precautionary reduction in the shrimp harvest.

One drawback of the current stock assessment models for northern shrimp is that they are not age(sex)-based, although the descriptions of the stocks do include discussion of the sex ratios and their expected impact on recruitment for the next year's population. This is partly due to the difficulty in directly determining age, a challenge which is in the process of being addressed (Jie et al., 2017).

Results of the current stock assessment models demonstrate best practice given the availability of data on biological conditions. As such, they reflect a biological approach to management using the volume measure of maximum sustainable yield (*MSY*) rather than a value measure of maximum economic yield (*MEY*) in decision-making. The stock assessment models place the stock at approximately 11% above the biomass level associated with maximum sustainable yield B_{MSY}, and current recommendations target continuing to keep biomass at or above this level (Burmeister and Kingsley, 2016a; Kingsley, 2016). This recommendation creates a population buffer to accommodate unforeseen changes – the volume of harvests could be sustainably higher if conditions remain stable but this is an uncertainty. B_{MSY} is the biomass (stock level) associated with the highest level of growth (and volume[biomass] of catch) that can be sustained year on year, and is frequently used as the target for fisheries management. Use of volume-based *MSY* to set target harvest quotas reduces the dimensions needed to provide management advice to quantity alone, rather than per-unit values and quantity, but it neglects the roles of price and cost, or in other words the human incentives and profits. Economists recommend use of *MEY*, where the volume of the fishery is converted to the revenue potential of the fishery and catch volumes are adjusted to reflect the costs (effort) of achieving that catch[2]. If done to best effect they will also include any external costs such as damage to benthic habitat or other species, or even climate, see for example Waldo et al. (2016).

Imagining a model including ocean acidification concerns

Adapting a cohort model for shrimp to a logistic function

As described above, age-based cohort models will be difficult for northern shrimp, and the logistic function has been successfully used in the formal stock assessment models based on Schaefer surplus production (Kingsley, 2016). The following discussion illustrates how a detailed cohort model might be calibrated to a logistic function that would enable consideration of ocean acidification impacts on the age and sex structures of the population, and therefore its dynamics.

Illustrative cohort model

A cohort model includes age groups in the stock assessments. This can potentially improve the ability to model changes in the stock behavior from environmental effects on different age groups. Existing knowledge of the shrimp populations is used here to illustrate what a cohort model for the fishery would look like in order to try and identify with some specificity how changes in ocean acidification might affect population dynamics. It is clear that cohort models are not yet a feasible modeling option to address these questions but that the exercise of describing such a model advances the state of knowledge about both known and unknown unknowns. This provides guidance for future research efforts.

A cohort model consists of a record of the number of individuals at a given age. As the shrimp have relatively short lifespans the cohort in this case is to be followed on a day-to-day basis. Northern shrimp undergo a sex change as part of their life cycle, which makes it relevant to model each sex. The model would comprise, for example, the population vectors Nm,t and Nf,t, where m,f indicate male or female sex and t indicates time (see Figure A4.18).

All individuals start as males and transition to female with a probability $p=p(t)$. The shrimp undergoes somatic growth (growth of the body, exclusive of the gametes) where the growth varies over the year (Bergström, 2000). Each sex has a different somatic growth function for length (l): $lm(t)$ and $lf(t)$. Burmeister and Kingsley (2014) have also established a weight-length relation $w=w(l)$ (equal for both sexes) which could be used for calibration.

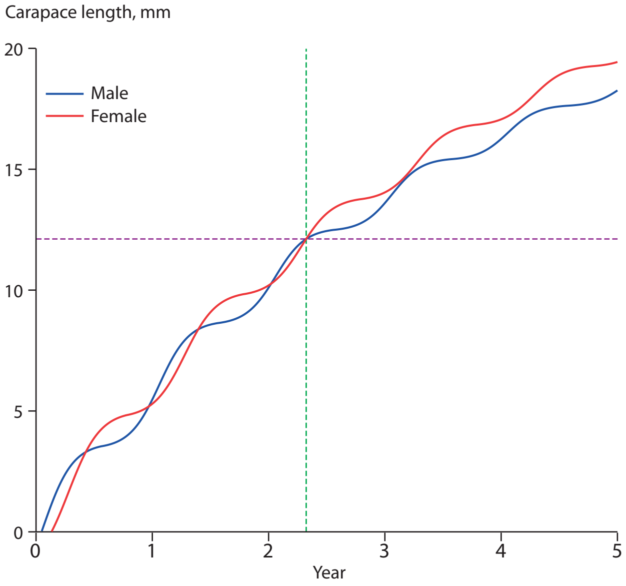

Figure A4.18 Somatic growth in northern shrimp by sex, adjusted from Bergstrom (1992) for shrimp off West Greenland. The green vertical line marks year 2 of body growth and the horizontal purple line shows that length at year 2 is about 12 mm. Note that an individual shrimp will only be one sex at a time. The graph should thus be read for an individual shrimp as the male function up until the transition and the female function after the transition, which is here modeled to occur at a little over 2 years.

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[2] The first, and still extremely clear, introduction to this concept was by Gordon (1954). This seminal contribution has now been cited more than 5000 times as economic principles become increasingly utilized in resource management.

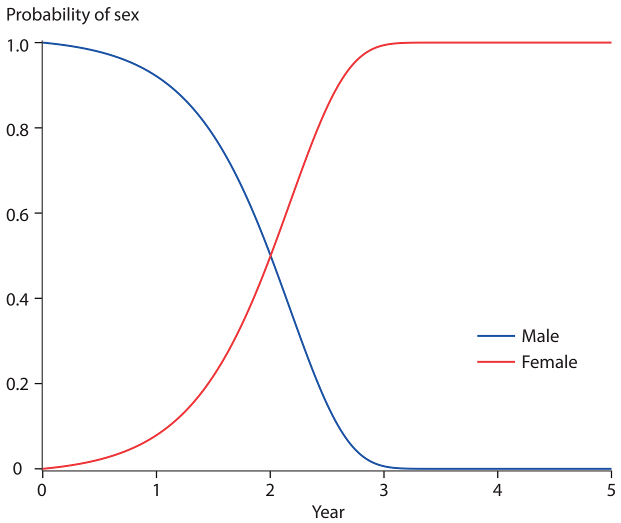

Figure A4.19 Transition probability function for the switch from male to female in northern shrimp.

Bergström (2000) parameterized a von Bertalanffy growth function, adopted with a sigmoid function for seasonal varying growth for *P. borealis* in a Swedish fjord, with much faster growth than for Greenland. Figure A4.18 illustrates the model when reparameterized for Greenlandic growth conditions.

The transition from male to female could be modelled as a probability of changing sex each day; the model uses a logistic function where the parameter *t*- shifts the function along the time axis. Figure A4.19 illustrates a possible transition probability function from male to female. However, as the model is not calibrated to Greenland specifically, the transitions may appear to be occurring slightly earlier than is actually the case. It may be calibrated in a completely specified model when fishery and natural mortality are calibrated.

Fishing and predation mortality

The population is exposed to two types of mortality: fishery mortality and cod predation mortality. The fishery mortality rate is *F* and the overall cod predation mortality rate is *M*.

The logistic function estimated for the shrimp off West Greenland for the Schaefer production model described in the Section *Current stock assessment model of the fishery* (Burmeister and Kingsley, 2015a; Kingsley, 2015) gives *MSY*=140.2. This includes natural, fishery and cod mortality. As the logistic (Schaefer) stock assessment model only considers the fishable stock, it is assumed that the cod predation included in *MSY* only accounts for cod predation of fishable stock. The cod predation is dependent on cod biomass and a functional response for each cod.

In adapting a cohort model to the logistic function, the cod predation factors in the current stock assessment model must also adjust for the younger stock. As small cod are more numerous than large cod, the mortality of cod is expected to scale with $w^{-0.25}$ where *w* is the weight of the shrimp. The fishery is only targeting shrimp with a length ≥17 mm.

Mortality from cod predation, or the annual consumption of shrimp by cod V_t follows from Burmeister and Kingsley (2015b) and can be derived from:

$$V_t = E_t \frac{V_{max} P^2_{t\text{-}1}}{P^2_{t\text{-}1} + P^2_{50\%}}$$

Where E_t is the cod stock biomass, V_{max} is the maximum consumption possible, and P_t, $P_{t\text{-}1}$ are the current and previous year's shrimp populations relative to B_{MSY}:

$$= \frac{B}{B_{MSY}}$$

If a steady state equilibrium is assumed, then $P_t = P_{t\text{-}1}$. Thus one translates the cod biomass into the losses of the shrimp and the rate of this loss, *M*.

The logistic function with functional response

A cohort model such as the above could then be calibrated to give the same results as estimated by Burmeister and Kingsley (2015a), that is, to give an *MSY* of 140.2 tonnes. This would provide a calibrated cohort model to the existing, well-functioning short-run population models. Theoretically this increases options for directly including effects of ocean acidification at different life stages of the shrimp into forecasting models of population and sustainable catch. The benefits of doing so, however, depend on the ability to forecast the effects of biogeochemical changes on life stages with enough certainty that the model is not overwhelmed by unknowns.

For calibration, it is necessary to find the number N_0 of hatched male shrimps and the fishing mortality rate F_{MSY} that give the *MSY*, given that the fishing mortality rate *M* also satisfies the functional response given a cod stock of 55 tonnes. In computation, it requires finding a root (hatched male shrimps) nested in an optimization (fishing mortality that achieves *MSY*) nested in a root finding (functional response to cod stock).

The *MSY* in the logistic (Shafer) model includes consumption by cod. The landed shrimp biomass (y-axis of Figure A4.20) shows the landed results of the fishing mortality (x-axis), given an effective cod biomass of 55,000 tonnes cod.

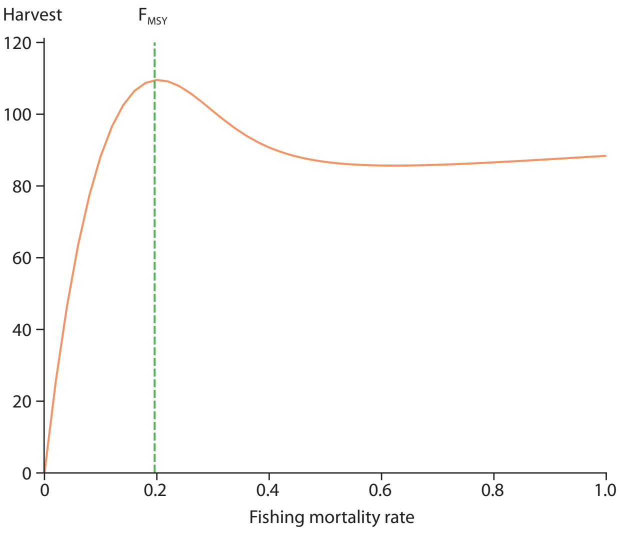

Figure A4.20 The landed biomass of northern shrimp given an effective cod biomass of 55,000 tonnes as a function of the fishing mortality (x-axis). The vertical line indicates F_{MSY}, where the highest volume of shrimp may be harvested repeatedly over time.

Table A4.3 Economic parameters for the West Greenland shrimp fishery.

	2006	2007
Harvest, tonne	55,305	70,675
Value, thousand DKK	562,970	679,470
Total private cost, thousand DKK	-487,283	-595,970
Private benefit, thousand DKK	75,687	83,500
Management (Home Rule) cost, thousand DKK	-17,089	-17,431
Net benefit, thousand DKK	58,598	66,069
Crew rent, thousand DKK	46,651	59,216
Net social benefit, thousand DKK	105,249	125,285
Rent per shrimp, DKK/kg	1.9	1.77

Maximum economic yield

With a calibrated cohort model as described above, fishing mortality, and age-structured concerns for sex composition of the population, the biological model of the fishery could respond to anticipated changes in rates of somatic growth; transition times for sex; the indirect effects of cod population increases (from temperature or other changes); change in the number of hatched male shrimp, for example as a consequence of change in forage; and the anticipated changes that come from price and cost require additional data input.

Christensen and Vestergaard (1993) built a bio-economic model of the Greenlandic shrimp fishery that includes careful delineation of rents as well as prices. Their model has been updated for the latest data available for the fleet (see Table A4.3). The average value per kg harvested over the two years is estimated to be 1.84 DKK.

More detailed data with which to estimate production or cost functions, which would relate levels of effort to catch (production) and costs for the shrimp fishery, and enable calculations of marginal benefit and cost, are not available. It is therefore not possible to estimate how cost changes with a change in stock, and so identify how to change effort or catch levels to maximize economic return. The best estimate for maximum economic yield (value) will then be maximum sustainable yield (volume). The preferred goal is to estimate the economic changes, and here economic yield at maximum sustainable yield is a good indicator.

The aim of a bio-economic model is to estimate changes in economic yield under predicted changes in the parameters of the shrimp. Even with a refined model (such as described above) in place and the resultant information about expected changes, it might still be the case that it is not possible to predict any change with certainty. One solution could be to try to compensate for the uncertainty by assuming a distribution of changes in the parameters and to run a Monte Carlo simulation to obtain a distribution of the predicted consequences. However, these assumptions of distributions of changes could only really be considered guesses (i.e., ignorance disguised as maths). This is especially true because the distributions of the uncertain outcomes are in many cases unidentified. It must therefore be concluded that the bio-economic modeling approach is at this stage currently a 'dead end' and as a result that the next steps must be to focus on these uncertainties to identify where research efforts might improve the applicability of such a model in future endeavors.

A4.7 Uncertainties and sensitivity assessment

> *Reports that say that something hasn't happened are always interesting to me, because as we know, there are known knowns; there are things we know we know. We also know there are known unknowns; that is to say we know there are some things we do not know. But there are also unknown unknowns – the ones we don't know we don't know. And if one looks throughout the history of our country and other free countries, it is the latter category that tend to be the difficult ones.*
>
> Donald H. Rumsfeld, 2002

Science is about organization of knowledge, that is, one purpose is to move the unknowns into the known category. In the Rumsfeld citation above, there are known knowns and known unknowns. From a science perspective, these should not be two discreet groups but rather a continuum; science is a process of bringing something from black to white, but all the shades of grey are also possible. A common observation is that for every new insight many more questions emerge. Seen in this way, what science does is not only to work on moving known unknowns towards the knowns category, but as importantly moving unknown unknowns into the known unknowns category. That is, by doing research in an area the first finding can be that things are more complex than originally thought.

As illustrated in Figure A4.1, there is scope for both known and unknown unknowns to be established in the causal links of the system for understanding the impacts of ocean acidification on the Greenland shrimp fishery. In this example, to predict the shrimp's reaction to ocean acidification it will first be necessary to predict the changes in chemistry, temperature and other environmental variables that the shrimp will experience. Also, because northern shrimp are positioned in the middle of an ecosystem and food web, eating plankton and in turn being eaten by predators such as cod, it will also be necessary to predict how the ecosystem as a whole will adapt to the changing environment.

Most of the cited research aimed at quantifying change in shrimp with respect to ocean acidification uses the experimental approach of changing one parameter (e.g., pH) and quantifying the difference compared to a control. What these experiments can examine is the capacity of individuals in present populations for physical adaptation. This can be seen as quantifying the short-term effect of that physical change, before any long-term effect of evolution can be considered; in general, any long-term effects of evolution are likely to mitigate impacts from ocean acidification but this is not certain. If individuals taken from one environment are placed in another environment they will probably on average do worse than in their original environment. But as there is diversity in the population some will do better than others in the new environment and by natural selection the population may adapt to the new environment. There are (uncertain) limits to what populations can adapt to.

The point is that the evolution may work to adapt species to a new environment, and the ability to adapt will differ from species to species. This may therefore also change the predator-prey and competitor relations in the community. The fate of one population not only depends on the ability to physically adapt and the ability to evolve, but also on the same abilities of other species in the community.

Modelling can take different approaches to address the complexity. At one end of the spectrum is a fully described and parameterized end-to-end modelling where everything from climate and biochemistry through ecosystem and human behavior is included. This is the desired approach here, and the form of its exposition. In practice, however, the interrelations of the known and unknown unknowns along the path, even without feedback effects from both ecological and economic adaptation render any results inconclusive. Efforts to determine consistent lower-lower bounds or upper-upper bounds to accommodate uncertainties suffer from uncertainty about the existence of unknowns and from an inability to predict even the direction of impacts in some cases, let alone their magnitude.

A simpler approach is a split model where chemistry, temperature and other environmental variables are modelled explicitly, followed by another model where the biology and fishery are modelled as an interacting system. The biological model in principle must be an ecosystem model, where the interactions among species are modelled. Present ecosystem models are not good at predicting changes in species dynamics even without environmental changes. Thus the best option for predicting changes in a single species may be to just model the species (shrimp) itself and then adjust the physical parameters accordingly. This creates results that more closely resemble scenario analyses, which do not impose probability assumptions on potential outcomes.

This section lists the components required for the simplest model to establish the impacts of ocean acidification on the Greenland northern shrimp, and includes information about where they currently sit on the known unknown scale. Geochemical changes are discussed first because they underpin the potential for biological and biogeochemical responses.

Geochemical changes

Change in CO_2 content in the atmosphere

The rate of change in CO_2 in the atmosphere will ultimately depend on human action, but here it is assumed that changes in CO_2 stem from forces external to the model and are represented by scenarios. The model results presented here are based on RCP8.5 (i.e., with no additional mitigation, namely 'business-as-usual'). Other scenarios (RCP6.0, RCP4.5, RCP2.6) include varying levels of mitigation (Moss et al., 2010). By starting from a 'business-as-usual' assumption about climate change policy and action, feedback effects from any policy actions that reduce CO_2 emissions should be negative, that is, they should decrease potential impacts. Such feedback effects are not considered here.

Change in ocean temperature

The change in ocean temperature will depend on changes in the consequences of CO_2 forcing. Modelling efforts for the Baffin Bay / Davis Strait region seem to be in agreement on an increase in sea surface temperature. The ESMs show a regionally uniform increase of about 0.2±0.1°C/decade for RCP8.5 (± indicates the inter-model standard deviation, Steiner et al., 2015).

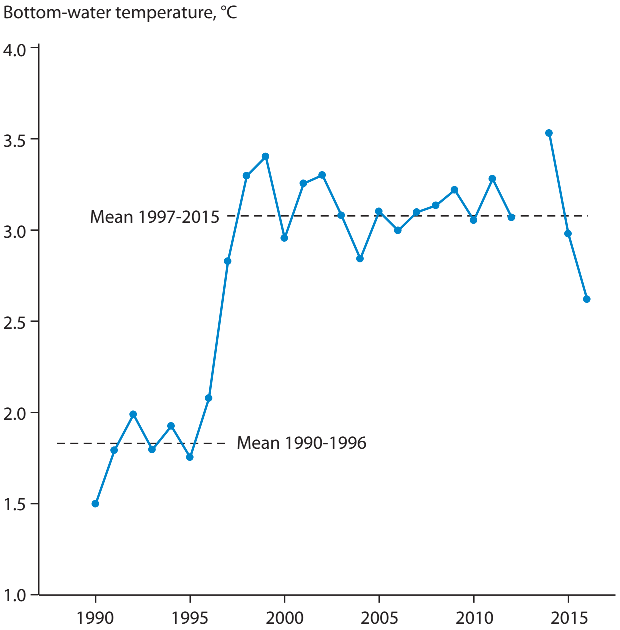

Figure A4.21 A shift in area-weighted average bottom temperature (Burmeister and Kingsley, 2016b).

There has indeed been a measured increase in bottom temperature in the fishing grounds since 1990 (see Figure A4.21). Although these changes vary by area (for detailed data see Burmeister and Kingsley, 2016b), overall there is a clear difference between average temperatures of the early 1990s and the later period.

This might have been expected to signal an increase in cod presence, but to date there is no significant correlation between the change in bottom temperature and the change in effective cod biomass (at only 0.1138; r^2=0.0662).

Change in ocean aragonite saturation levels, calcium carbonate saturation levels, and pH

The levels of change in ocean aragonite saturation, calcium carbonate saturation, and pH will depend on changes in CO_2 and resulting changes in climate, including ocean temperature. Model projections suggest that a mid-range emission scenario with mitigation (RCP4.5) would reduce the rate of acidification (a pH of 7.9 is reached about 25 years later under RCP4.5 than RCP8.5). However, the emergence of undersaturated surface waters, which is projected to occur within the next decade, differs little between scenarios (Steiner et al., 2014).

Biological and biogeochemical responses

Once the impacts on geochemical forces have been predicted, their potential impacts on the life cycle of the northern shrimp can be predicted. The literature identifies the juvenile parts of the life cycle for many invertebrates as being the most sensitive to ocean acidification and related climate change drivers (Byrne, 2011).

Change in survival rate

In the short term, there will be a range of physical adaptations affecting survival rate. In the long term, survival rate changes will include adaption through evolution. There is some indication of reduced survival rates in the short term (Dupont et al., 2014). If a population is adapted to a certain environment, any change in environmental parameters may cause increased mortality. This is part of the process of the population adapting to the new environment. Adaptation through evolution is the driver that creates resilience in the long run. Without many generations of experiments it is hard to predict the limits to what evolution can accomplish with respect to adaptation.

Change in growth rate

In the short term, there is evidence that shrimp are able to modify physiological processes such that they are able to persist under ocean acidification (see Table A4.1). Over the long term, when including adaption through evolution, results suggest that northern shrimp have relatively well-developed mechanisms to counteract CO_2-induced acidosis (Hammer and Pedersen, 2013). The mechanism requires increased regulation and is therefore costly in terms of energy. It does not seem to be of an order that can be measured directly on the growth rate. Meanwhile, increased temperature within the ranges predicted seems to favor increased growth rates.

Another complication is that the growth rate is not only affected by the physiological capacity of the organism but also by food availability. The prognosis for primary production seems to be uncertain to the point where both increases and decreases are reasonable predictions, but probably with some locations showing increases and others showing decreases. Based on this high level of uncertainty, knowledge of long-term evolutionary responses cannot be predicted. Furthermore, there may be a spatial divergence of effects. Any such divergence will mean impacts differing by community such that spatial monitoring and adaptability of governance decisions for the fishery will improve potential impacts on human populations.

Change in the sex transition mechanisms

In the short term (range of physical adaptation) and over the long term (including adaption through evolution), shrimp sex ratios are controlled by the timing of the transition from male to female. Any change in the environment may affect the transition time but it is not known for certain if this is the case. For secure reproduction rates, a population must have sufficient numbers of each sex present. The sex ratio is not only a function of transition timing but also of longevity. From this it follows that survival rates, growth rates, fishery choices such as size restrictions, and many other factors can influence the sex ratio. These influences are unknown.

What is known is that there is large variation in the timing of when individuals experience sex transition, and that part of the variation is caused by genetic variation. Over the long term, it can therefore be assumed that evolution will maintain the contemporary sex ratio by adjusting the sex transition time; this being when there is an equal chance of reproduction whether the shrimp transits a little before or a little after the mean transition time. What is not known is the final outcome in terms of transition timing, which prevents improvements to management advice regarding parameters such as optimal harvest age. The only way to include this within a model would be to assume the evolutionary equilibrium.

Change in reproduction

Recruitment is considered in a fishery cohort model through parameterizing the number of recruits that enter the first cohort. That is, recruitment includes fertile eggs, the larva stadium and the first benthic life in one stage of the model. Survival rates through these early life states are also important. As the surface:mass ratio is larger for these small lifeforms, they will be more vulnerable to changes in the environment. From this alone, higher mortality can be expected in the short term (physical adaption period). Higher mortality also means that selection is stronger and evolutionary changes could be expected to act upon the physical properties of the shrimp such that they respond effectively to the environmental change. How fast and to what extent are unknowns. The best advice for any modeling effort is to expect higher mortality and thus secure recruitment by an oversupply of eggs. That is, choose to have a conservative harvest policy as the northern shrimp fishery is mainly taking (the larger) females (often with eggs).

Changes at the ecological/community level

Impacts from predation and mortality occur and are measurable at the community level. Biogeochemical impacts affect individuals in the short term and populations over the long term. In turn, these short- and long-term impacts will filter through the ecosystem at community level altering the population dynamics upon which harvest decisions are made. Because cod are also harvested, human interactions play multiple roles in determining outcomes within this fishery. While information is available about past conditions the variability in these data stems from both human and natural impacts and this makes it difficult to predict long-term changes arising from external forcing caused by ocean acidification and related climate factors.

Change in predator conditions

There are some indications that warmer water may favor cod or other predators. Current stock assessment models in the fishery account for cod predation, such that if higher cod populations are identified, stock assessments can adjust. The modelling may have to be revised if changes expand beyond simple increases in population, such that the models include, for example, changes in depth overlaps of the predator and prey that in turn affect predation rates for a given population. Current assessments include an 'effective' cod biomass – that is, a measure of the cod biomass expected to be at the right depth in the right place to predate on the shrimp. Thus, as ecosystem changes shift temperatures, the cod and shrimp will change both location and depth, which alters not only their interactions but may also alter fishing behavior and fishing mortality for either or both species. The direction of these changes is not currently predictable, nor is the magnitude

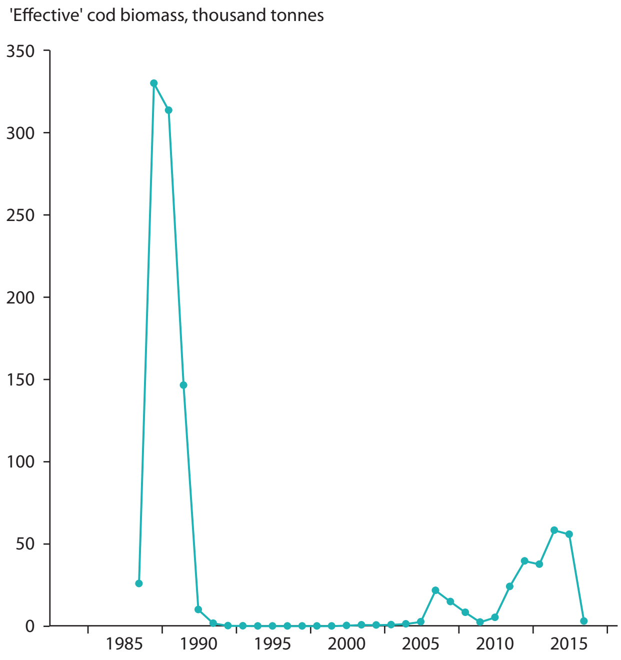

Figure A4.22 Index of the 'effective' cod biomass in NAFO Divisions 0A and 1. This index measures overlap between the stocks of Atlantic cod and northern shrimp (i.e., the amount of cod biomass expected to be at the right depth in the right place to predate on the shrimp) (Burmeister and Kingsley, 2016b).

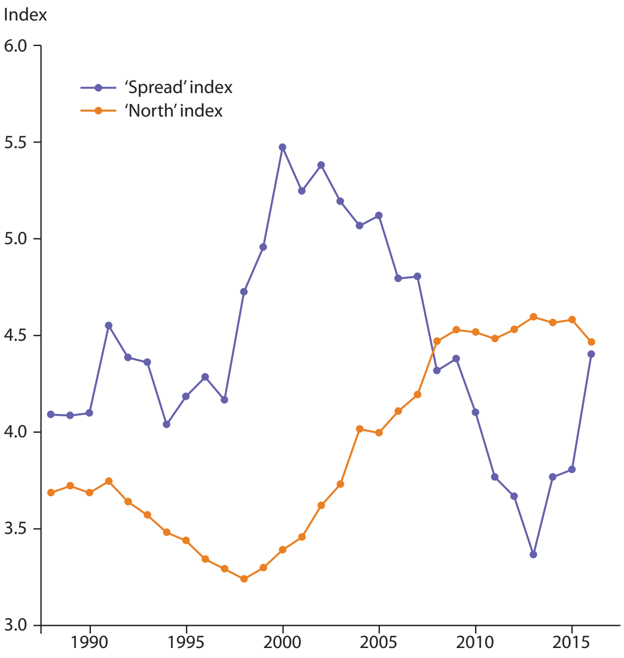

Figure A4.23 *Pandalus borealis* off West Greenland: indices of distribution and location of shrimp biomass in the West Greenland trawl survey (3-point moving averages) (Burmeister and Kingsley, 2016b).

of change. The historical record (see Figure A4.22) shows significant potential shifts as well as volatility. Continued observation is the best recommendation available to ensure active, adaptive management.

Change in food availability

The biogeochemical analysis suggests that primary production may increase or decrease, or increase in some locations while decreasing in others. Food availability is not only a function of primary production but also of change within the rest of the ecosystem, environmental change, and the effect of the fishery. Thus, it is unclear whether food availability for northern shrimp will increase or decrease, or whether locational shifts in food availability will redistribute fishery productivity and change the distribution of returns to the fishery even with constant populations.

Greenland northern shrimp habitat

Change in the distribution of northern shrimp is a function of the shrimp's aptitude for physical adaptation, of change within the rest of the ecosystem, environmental change, and the effect of the fishery. Even under contemporary environmental conditions it is not really known how the distribution of the shrimp population may vary from year to year. The Greenland Institute of Natural Resources tracks the spatial spread of northern shrimp in two dimensions. The 'north' index tracks how much of the population is captured north of 69°30'N, while the 'spread' index tracks the distribution of the population across NAFO Divisions 0A and 1 (Figure A4.23).

The shrimp population may also respond to such drivers by shifting their average depth (Figure A4.24). For example, the graphic shows a significant relative movement of shrimp biomass between 200–300 m and 300–400 m. After a period of constant or declining biomass at all depth strata between 2003 and 2013 there appears to be some growth in biomass at the 200–300 m depth stratum in particular. Such shifts must be considered as a combined potential response and a source of additional change for both the ecosystem and the fishery.

Variability in the 'north' and 'spread' indices is expected to be correlated with both ecological and human drivers. These shifts will in turn affect the economic costs of the fishery and potentially the distribution of returns, particularly if there is movement from coastal to offshore waters or vice versa. Flexible institutions for quota distribution should be maintained to reduce the likelihood of transferring significant levels of quota away from the coastal fleet to offshore interests. Since the shrimp fishery represents such a major element of the Greenlandic economy these concerns are well incorporated into decision-making.

Economic changes

Economic factors will directly influence the expected ecological and social outcomes in the fishery, with feedback effects to models and fisheries assessments. The economic factors may be external, such as changes in price, or may more directly reflect changes due to ocean acidification and other climate factors. Direct changes and their uncertainties are the focus of this section.

Changes in taste

Just as ecosystems adapt so will human tastes and preferences. While some evidence (Sam Dupont et al., 2014) from small

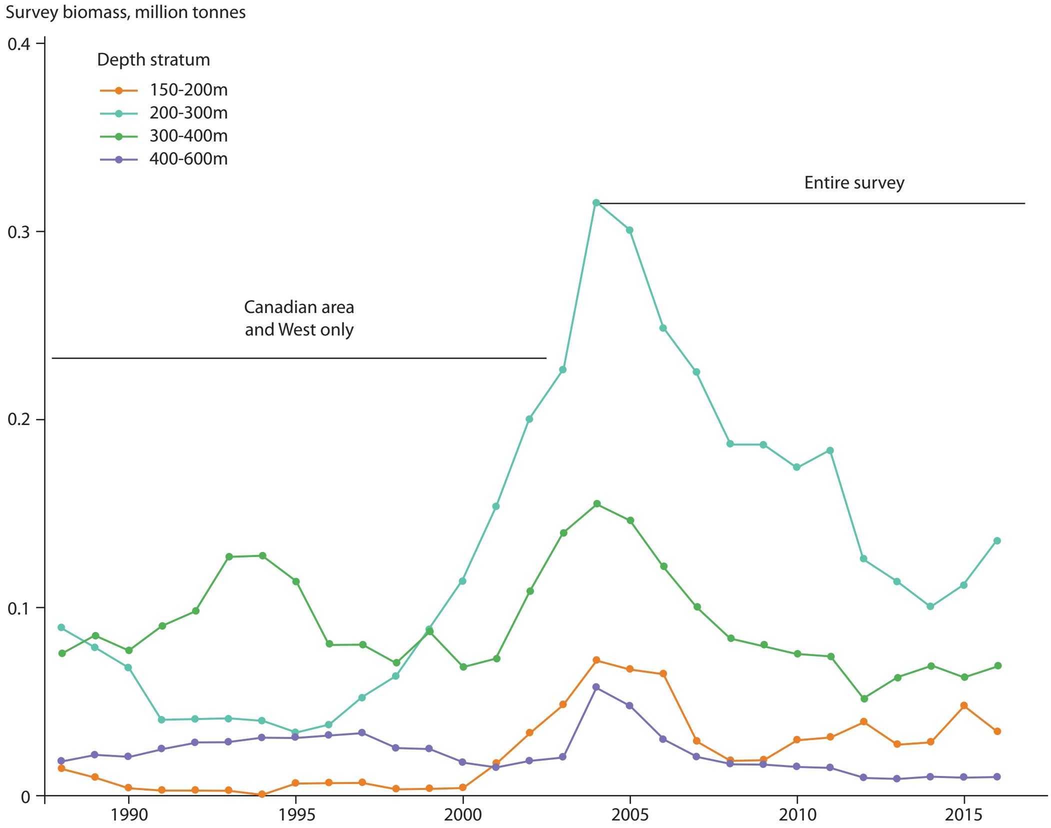

Figure A4.24 Survey biomass estimates by depth stratum for *Pandalus borealis* off West Greenland. Prior to 2003, only the Canadian area and the West regions; after 2003 the Northern area as well as Disko Bay and Vaigat entered the survey (Burmeister and Kingsley, 2016b).

surveys of human participants shows that ocean acidification may adversely affect current tastes, human responses may include a lower willingness to pay (affecting price and reducing demand) or switching to other foods (reducing demand). However, if ocean acidification is also changing the taste of shrimp's nearest substitutes there may be no net redistribution of consumption. Furthermore, rising world population and increasing wealth are driving a demand for protein, which may outpace any reductions from declining quality. Prices for Greenland northern shrimp have roughly doubled since 2010 (Figure A4.25).

Figure A4.25 Index of shrimp price. Data source: Statistics Greenland.

Technological development

Economists face difficult challenges in building models that can include technological change and this has been an under-researched area in resource economics and fisheries management. Squires and Vestergaard (2013) explored the implications of this for stock assessments and subsequent management. They found that when considering technical change, "steady state equilibria are spurious and a costly distraction to policy formation and impose costly burdens on fishing industries". Because technical change lowers costs, new balance must be continually established between the net benefits of a shrimp caught today and one caught in the future. These shifts in the intertemporal benefits create feedback effects that play out in both population dynamics and industry structure.

Technical change may also induce spatial shifts in industrial activities that affect communities in different ways. The expected challenges are similar to those already discussed regarding spatial shifts in biological productivity; there may be winners in some locations from increased access to shrimp for processing while others may suffer from reduced opportunities. There is more potential for human intervention to moderate these impacts, however, and they should be explicitly considered in regulation. The West Greenland shrimp fishery already does this to an extent by sharing the quota between the coastal and offshore fleets.

A4.8 Conclusions

This case study describes an attempt to build an end-to-end model to identify how ocean acidification could affect socio-economic outcomes in the West Greenland shrimp (*Pandalus borealis*) fishery. This includes an overview of how ocean acidification might impact the bio-geochemical conditions of the fishery and how the fishery sector might interact with these changes. Current tools for managing the fishery include state-of-the-art stock assessment models, but these do not currently provide many entry points to include change based on ocean acidification. In attempting to build a cohort model for the shrimp stock it is clear that the weight of economic and ecological uncertainties is too great for the model. Predicting change in the biology, ecology, and economy of West Greenland shrimp due to ocean acidification or other climate change impact remains difficult.

However, this work does move the science forward in several ways. First, it highlights where the gaps in scientific knowledge that are most likely to matter may lie. These include both short- and long-term changes in biological and ecological conditions for the fishery driven by ocean acidification as well as the likelihood of the ocean acidification occurring in ways that impact the fishery. Furthermore, this work includes a consideration of human responsiveness. Human reactions to changes in quality and quantity of shrimp will matter both on the demand side (e.g., taste of the shrimp or prices for substitutes) and on the supply side (e.g., when, where and how much to fish for shrimp). In addition, supply-side impacts, including technological change, may have spatial impacts on community dynamics that affect long-term wellbeing for Greenlandic populations in many ways, creating winners and losers from changes in access to and volume of the harvest.

Moving from biogeochemistry to management needs caution, but caution should not equal dismissal of the opportunity to increase understanding of both known and unknown unknowns. This is the focus of Section A4.7. The many uncertainties mean there are no immediate options for 'getting ahead of the issue', that is, to pro-actively change management in ways that definitively improve socio-economic outcomes and the sustainability of the fishery. Furthermore, being able to prevent the fishery from experiencing ocean acidification is unlikely. However, in examining the recent history of the fishery (Section A4.5) it is clear that current management is occurring at a high level of adaptability and with an ecosystem-based approach in place. This comprises direct management efforts that include an assessment model that includes predatory cod interactions (Bayesian surplus production models); annual surveying and quotas; annual update of the model; and annual recommendations.

Current management also includes indirect approaches that address broader socio-economic concerns including onshore processing requirements, distributional requirements between coastal, offshore and international interests built into quotas, and limits to quota concentrations that would also increase wealth disparities. In short, the fishery is well managed for its current goals. As a result, the best recommendations do not concern changes in harvesting that might arise from a detailed integrated bio-economic model of the effects of ocean acidification and other climate change drivers on the shrimp fishery, but concern means to more broadly manage change under uncertainty. The recommendations are as follows:

- *Monitor and be adaptive, and cooperate more with Canada.* Monitoring spatial changes in particular may assist in separating out the co-factors of change. If temperature increases are driving species north, but pH changes are affecting species from the northwest to the southeast, then visible changes that can be identified as stemming from one direction to another will be more informative. If such spatially delineated scientific evidence becomes available, more pro-active adaptation to the impacts on the shrimp populations can include negotiation of quota allotments between Greenlandic and Canadian waters as well as internally between the coastal and offshore fleets. This can provide increased stability in the face of impending change.
- *Build economic resilience.* The northern shrimp fishery is a significant part of the Greenlandic cash economy and many depend on its continued prosperity at both the national and local level. Ocean acidification and other climate drivers may act through both ecological and economic forces to concentrate harvests and profits as well as to change the overall productivity of the fishery. Changes in the overall productivity of the fishery will affect the direction of fishery revenues and their importance in society as a whole; diversification of economic activity can reduce national impacts from negative change. At the same time, changes in the concentration of the harvest and its profits may affect the distribution of impacts from change induced by climate drivers. These local impacts can also benefit from economic diversification, in addition to careful consideration of negotiations made to share stocks. At both the national and local level, maintaining and enhancing flexible onshore physical and human capital investments are key elements of increased economic resilience. Forms of physical capital investments include multi-species processing; forms of human capital investments include increased education and job training outside of fisheries.
- *Invest in understanding market impacts and/or genetic adaptation.* The many scientific uncertainties in understanding the potential economic effects of ocean acidification require investment to resolve. The productivity, or benefit, of the research should be weighed against the costs of acquiring the knowledge. With so many uncertainties ranging from the biological and ecological to short- and long-term economic change, there may be some research paths that afford significant gains at relatively low investment compared to other options. For example, it is important to understand whether any changes in quality that might come to the shrimp through ocean acidification will matter to end users. In other words, can consumers identify differences in the products and will those differences affect their willingness to pay for the products? Answering this question does not have to wait until there are actual changes in the available product, as laboratory capabilities can create simulated impacts that reflect potential ocean conditions far into the future. Similarly, laboratory work can help identify expected trends in genetic adaptation under controlled conditions. While these may not reflect the full

ecological outcomes that will occur as climate drivers change Baffin Bay they can point the direction of field study toward avenues with the highest potential return for providing information that improves management of the fishery and subsequent human wellbeing.

In summary, the state of scientific information regarding how ocean acidification and climate drivers might impact the socio-economic outcomes of the West Greenland shrimp fishery is insufficient to improve on existing bio-economic modeling and resultant stock assessment models and governance. What can and should be done is to invest in decreasing both short- and long-term uncertainties, not only for biological and ecological interactions but also for economic and social ones, and in building economic resilience to cope with ecological change.

The case of the West Greenland shrimp fishery is a relatively simple one, where the main use of the shrimp by the Greenlandic producers is to raise cash and the cultural impacts are not expected to be significant. Furthermore, the bio-geochemical and ecological connections are not anticipated to be complex due to the relatively simple ecological interactions and life-cycle opportunities for change. In considering this analysis in the broader context of impacts on Arctic communities, the detailed account of the lack of evidence even for this initially simple-seeming case shows the strong need for much more integrated assessment.

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Annex 5

Ocean acidification risk assessment for Alaska's fishery sector

This annex contains a reprint of the article *Ocean acidification risk assessment for Alaska's fishery sector*, by J. T. Mathis et al., published in Progress in Oceanography, Volume 136, Pages 71-91. August 2015.

This case study examines the threat posed by ocean acidification and warming to the communities and economies of Alaska that depend upon the highly productive Alaskan fisheries. In situ observations of ocean acidification and warming, and the known species-specific responses to ocean acidification as well as the patterns of human dependence on marine resources, were used to determine risk and vulnerability of commercial and subsistence harvests regionally across Alaska. Impacts in these regions were also projected into the future based on climate scenarios developed by the Intergovernmental Panel on Climate Change (IPCC, Fifth Assessment Report). Results indicate that highly fishery-dependent communities in the southeast and southwest of Alaska that have relatively low incomes and few alternative sources of employment are highly vulnerable and so are likely to face the greatest risk to ocean acidification and global climate change. This integrated observing-modeling-economic impacts study is an important means for evaluating the human impacts of ocean acidification and for developing resilience and adaptation strategies. While a reduction in global carbon dioxide emissions is an essential long-term pathway to mitigating ocean acidification, Alaska (and more broadly the Arctic region as a whole) is already facing dramatic changes in environmental conditions and so should develop adaptation strategies as an urgent priority, particularly strategies that can be implemented immediately.

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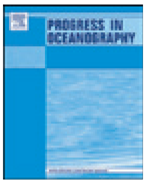

Ocean acidification risk assessment for Alaska's fishery sector

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ABSTRACT

The highly productive fisheries of Alaska are located in seas projected to experience strong global change, including rapid transitions in temperature and ocean acidification-driven changes in pH and other chemical parameters. Many of the marine organisms that are most intensely affected by ocean acidification (OA) contribute substantially to the state's commercial fisheries and traditional subsistence way of life. Prior studies of OA's potential impacts on human communities have focused only on possible direct economic losses from specific scenarios of human dependence on commercial harvests and damages to marine species. However, other economic and social impacts, such as changes in food security or livelihoods, are also likely to result from climate change. This study evaluates patterns of dependence on marine resources within Alaska that could be negatively impacted by OA and current community characteristics to assess the potential risk to the fishery sector from OA. Here, we used a risk assessment framework based on one developed by the Intergovernmental Panel on Climate Change to analyze earth-system global ocean model hindcasts and projections of ocean chemistry, fisheries harvest data, and demographic information. The fisheries examined were: shellfish, salmon and other finfish. The final index incorporates all of these data to compare overall risk among Alaska's federally designated census areas. The analysis showed that regions in southeast and southwest Alaska that are highly reliant on fishery harvests and have relatively lower incomes and employment alternatives likely face the highest risk from OA. Although this study is an intermediate step toward our full understanding, the results presented here show that OA merits consideration in policy planning, as it may represent another challenge to Alaskan communities, some of which are already under acute socio-economic strains.

Introduction

Marine environments around the world are now subject to unprecedented pressures resulting from human development, including increases in temperatures and atmospheric carbon dioxide (CO_2) concentrations, changes in terrestrial runoff, and intense exploitation of resources (Doney, 2010; Halpern et al., 2008). In Alaska (Fig. 1), highly productive commercial and subsistence fisheries are located in regions projected to experience rapid transitions in temperature, pH, and other chemical parameters, crossing distinct geochemical thresholds beginning this decade (Fabry et al., 2009; Steinacher et al., 2009; Mathis et al., in press; Cross et al., 2013). Ocean acidification (OA), the term used to describe the progressive decrease in marine pH and carbonate ion concentration driven by the uptake of anthropogenic CO_2, is a global phenomenon with localized effects on marine species. These effects are predominantly negative, although there is some variability within species groups (Barton et al., 2012; Kroeker et al., 2013a; Whittmann and Pörtner, 2013). Many of the marine

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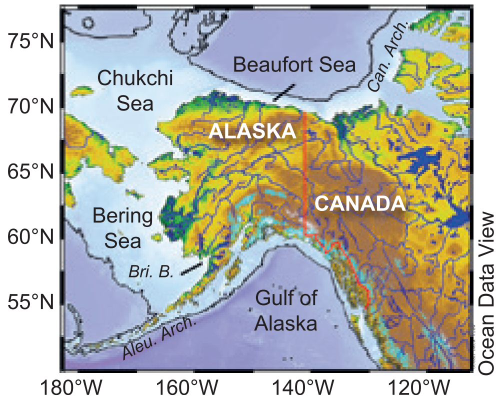

Fig. 1. Map showing the location of the major continental shelf seas around Alaska.

groups that are most intensely affected, such as mollusks and other shellfish, contribute substantially to Alaska's highly productive commercial fisheries and traditional subsistence way of life. Unfortunately, end-to-end assessments of how changes in seawater chemistry could affect key resources for specific human communities are limited in both scope and geographic coverage (Brander et al., 2012; Cooley et al., 2009; Cooley and Doney, 2009; Narita et al., 2012), and there has been no specific focus on Alaska or any other high-latitude region to date. To address this critical knowledge gap, we synthesized natural and social science data to assess the risk OA poses to Alaska's fishery sector.

Living marine resources are a critical part of Alaska's natural wealth portfolio that support a range of industries and activities, including commercial and subsistence fishing, tourism, and natural resource extraction. The revenue and protein from these sources provide economic and nutritional benefits reaching far outside the state's boundaries, to the U.S. Pacific Northwest and beyond. The state's 33,000 km coastline is 50% greater than the rest of the U.S. shoreline combined and produces about half the total commercial fish catch in all U.S. waters. The commercial fish catch also helps maintain the U.S. balance of trade on the global market. Alaska's commercial harvests had an estimated wholesale value of $4.6 billion and supported almost 90,000 full-time-equivalent jobs in the state in 2009 (Northern Economics, Inc., 2011). At the same time, the sport and personal fishing industry supported another 16,000 in-state jobs, and $1.4 billion of angler spending (Southwick Associates, Inc. et al., 2008). Fishing-related tourism yields over $300 million a year in revenue for Alaska, and makes up approximately half of the state's total economic income from tourism (Southwick Associates Inc. et al., 2008). Moreover, approximately 17% of the Alaskan population, roughly 120,000 people, depend on subsistence fishing for food, with 95% of households participating in subsistence activities using fish, and 83% harvesting fish. These activities are central to many cultural customs, and additionally important sources of employment and nutrition (Fall, 2012), with two-thirds of the entire state population living along the coast (U.S. Census Bureau, 2011). For example, the Bering Sea directly or indirectly provides over 25 million pounds of subsistence food for Alaska residents, primarily Alaska Natives in small coastal communities.

Ocean acidification near Alaska

Since the pre-industrial era, human activities have increased the atmospheric CO_2 concentration by about 40% to values now at 400 ppm, which is higher than at any point during the last 800,000 years (Lüthi et al., 2008). Meanwhile, the ocean has absorbed more than 25% of the total emitted anthropogenic CO_2 (Feely et al., 2013; Sabine and Feely, 2007; Sabine and Tanhua, 2010), helping to offset some of the atmospheric consequences of humanity's waste emissions. The oceanic uptake of CO_2 triggers a series of well-understood reactions in the surface ocean that has profoundly changed seawater chemistry around the world (e.g. Doney et al., 2009; Fabry et al., 2008; Feely et al., 2004, 2008, 2009; Orr et al., 2005). This mechanism of change has already reduced the global surface ocean pH by about 0.1 units (e.g. Byrne et al., 2010; Feely et al., 2004), making the ocean 30% more acidic than in pre-industrial times. Carbonate ions (CO_3^{2-}) naturally found in seawater partially neutralize this reaction and slow the decline in pH. However, this buffering mechanism depletes the seawater of CO_3^{2-}, which makes it more difficult for organisms like mollusks and corals to create and maintain their hard shells and skeletons. The progression of OA is often discussed in terms of the "saturation state" (Ω) of calcium carbonate minerals ($CaCO_3$), which is a measure of the thermodynamic potential of a mineral to form or dissolve. When the Ω for aragonite (Ω_{arag}) and calcite (Ω_{cal}) are below 1.0, the water is corrosive to $CaCO_3$ minerals. A comprehensive review of OA chemistry can be found in Gattuso and Hansson (2011).

High-latitude oceans, like those around Alaska (Fig. 1), have naturally low CO_3^{2-} concentrations and are thus considered to be more vulnerable to the impacts of OA on shorter timescales (Fabry et al., 2009), because additional losses of CO_3^{2-} from OA represents a much greater proportional change to the system. Waters circulating along the coastline of Alaska are derived from CO_2-rich waters that are upwelled in the North Pacific, where anthropogenically induced pH changes have already been directly observed (Byrne et al., 2010). As these waters flow generally northward into the Bering Sea, with some eventually entering the Arctic Ocean, low sea surface temperature and increased solubility of CO_2 promotes naturally low CO_3^{2-} surface concentrations (Key et al., 2004; Orr, 2011; Orr et al., 2005). Uptake of anthropogenic CO_2 further reduces the surface CO_3^{2-} concentrations, pushing the high-latitude waters closer to the threshold of undersaturation with respect to aragonite (Mathis et al., 2011a). Waters around Alaska are also subject to regional physical and biological processes that exacerbate the progression of OA by additionally decreasing pH and CO_3^{2-}, or increasing the partial pressure of CO_2 (pCO_2).

In the western Arctic Ocean, which encompasses the Chukchi and Beaufort Seas (Fig. 1), potentially corrosive waters (Ω_{arag} as low as 0.5 and Ω_{cal} as low as 0.9) are found in the subsurface layer of the central Canada basin (e.g. Jutterström and Anderson, 2010; Yamamoto-Kawai et al., 2009), on the Chukchi Sea shelf (Bates et al., 2009; Mathis and Questel, 2013), and in outflow waters on the Canadian Arctic Archipelago shelf (Azetsu-Scott et al., 2010). In the Chukchi Sea, waters corrosive to $CaCO_3$ occur seasonally in the bottom waters due to the combination of natural respiration processes and the intrusion of anthropogenic CO_2 (Bates et al., 2009; Mathis and Questel, 2013). Seasonally high rates of summertime phytoplankton primary production there drive a downward export of organic carbon that is remineralized back to CO_2, which in turn increases the pCO_2 and lowers the pH of subsurface waters. The seasonal biological influence on the pH of subsurface waters amplifies existing impacts of OA (Bates et al., 2013; Mathis and Questel, 2013). Aragonite undersaturation has been observed in bottom waters of the Chukchi Sea in July, August, September, and October (Bates et al., 2009, 2013; Mathis and Questel, 2013).

Unlike the Chukchi Sea, the Beaufort Sea shelf (Fig. 1) is relatively narrow with a limited physical supply of nutrients (e.g. Carmack and Wassmann, 2006). Rates of phytoplankton primary production over the shelf have been estimated at $\sim$6–12 g C m^{-2} yr^{-1} (Anderson and Kaltin, 2001; Macdonald et al., 2010), compared to $\geqslant$300 g C m^{2} yr^{-1} (i.e. Macdonald et al., 2010; Mathis et al., 2009) in the Chukchi Sea. Although respiration of this small amount of organic matter at depth is not likely to lower subsurface pH markedly, OA in the Beaufort Sea may nevertheless worsen due to loss of Arctic sea ice, as the ice dampens the transfer of wind energy and limits upwelling. During an observed upwelling event in the Beaufort Sea, upper halocline water, replete in CO_2 and undersaturated in aragonite, reached the surface and moved all the way inshore along the Beaufort shelf, covering thousands of square kilometers (Mathis et al., 2012). Although some level of storm-driven upwelling is typical in this region, especially in autumn, land-fast as well as pack ice has historically returned before major late-autumn storm systems begin to pass through the region. In recent years, the western Arctic has seen an unprecedented loss of both sea ice extent and volume, so the shelves are staying sea ice-free longer each year through September and October while storm frequency and intensity reach their annual peak. In the future, the Beaufort shelf is likely to be persistently, if not continually, exposed to waters that are undersaturated in aragonite as sea ice cover continues to diminish under warming conditions (Mathis et al., 2012).

Much like the Chukchi Sea, the Bering Sea (Fig. 1) experiences seasonal variability in primary production and remineralization of organic matter (Cross et al., 2012; Mathis et al., 2011b), which both control the carbonate chemistry of the water column (Cross et al., 2013). Biological production decreases the $p\mathrm{CO}_2$ at the surface (Bates et al., 2011; Cross et al., in press) and increases Ω in summer (Mathis et al., 2011b). The $p\mathrm{CO}_2$ can range from 150 to 400 μatm in the surface mixed layer, while Ω_{arag} oscillates between an annual maximum of 3.5 and a minimum of 1.2. The only surface locations where aragonite has been observed to be undersaturated were where sea ice melt or river runoff predominated, both of which are low in total alkalinity (TA) relative to marine waters (Mathis et al., 2011b). Export and remineralization of large quantities of organic matter from surface blooms sharply increases the $p\mathrm{CO}_2$, lowers pH, and decreases Ω near the bottom, particularly in summer and autumn months. Moored sensors near the bottom showed that $p\mathrm{CO}_2$ levels exceed 500 μatm by early June and remain high well into the autumn, indicating that bottom waters are likely continuously undersaturated in aragonite for several months each year (Mathis et al., in press), primarily due to natural respiration. However, the extent, duration, and intensity of these undersaturation events will likely increase as anthropogenic CO_2 inventories continue to rise in the water column and average Ω declines. The timing and duration of these undersaturation events could be significant for the development of larval and juvenile calcifiers in the region (e.g. Long et al., 2013a,b).

Unlike the vast continental shelf regions to the north, the Gulf of Alaska (Fig. 1) does not have seasonal sea ice cover. However, it receives both low-alkalinity water (and hence lower Ω) from glacial runoff (Reisdorph and Mathis, in press; Evans et al., 2014) and upwelling of waters that are rich in CO_2 and undersaturated in aragonite from the deep Gulf of Alaska (Evans et al., 2013). Throughout most of the year, alongshore winds create a downwelling environment that keeps deeper water from penetrating onto the shelf. However, in summer these winds relax, allowing the waters that are undersaturated in aragonite to penetrate the inner shelf, causing the saturation horizon for aragonite to become as shallow as 75 m (Evans et al., 2013). Although the narrow continental shelf of the Gulf of Alaska is more than three times as deep as the Bering and Chukchi shelves, there is still a considerable remineralization of organic matter at depth that further drives a reduction in pH and Ω in the bottom waters.

Alaska's marine organisms and ocean acidification

OA appears to act more strongly on certain species and types of organisms than others (Kroeker et al., 2013a; Ries et al., 2009; Whittmann and Pörtner, 2013; Table 1). More calcifying organisms than non-calcifiers clearly exhibit significant negative responses (Kroeker et al., 2013b; Long et al., 2013a,b), and lower pH environments alter ecosystem composition toward dominance by non-calcifying organisms (Hall-Spencer et al., 2008; Wootton et al., 2008). Mollusks appear to be the calcifying group most negatively affected by OA. However, mollusks represent a very small fraction of Alaska's marine harvests, and the specific OA responses of most species harvested in Alaska, mollusks and others, have not yet been fully studied. We must therefore infer responses from studies on similar mollusk species (Table 1) and on meta-analyses of mollusks overall (e.g. Kroeker et al., 2013a), which suggest that it is more likely than not that harvested mollusk species in Alaska will experience negative effects from OA.

The biological OA responses of only two commercially important Alaskan crustacean species have been directly studied. Both red king crab (*Paralithodes camtschaticus*) and Tanner crab (*Chionoecetes bairdi* and *C. opilio*) species exhibited negative responses to high-CO_2, lower-pH water (Long et al., 2013a,b). Growth of red king crab was slowed and molting success decreased in waters with a pH of 7.8, and crabs died in highly acidified conditions (pH = 7.5). A similar pattern was observed for Tanner crabs in waters with a pH of 7.5, although they had a higher survival rate. Studies on crustaceans from other locations also show negative effects on core physiological processes in response to decreased pH (Pane and Barry, 2007; Walther et al., 2010). This is particularly important in the early stages of development, when organisms tend to be more sensitive. In addition, species inhabiting cold, Arctic waters show narrower thermal tolerances in response to higher CO_2 levels (Walther et al., 2010). In several high-latitude species, negative responses to decreased pH are particularly strong when combined with other stressors such as increasing temperature (Enzor et al., 2013; Strobel et al., 2012). Moreover, deep-water species may be less tolerant to changes in pH due to the natural stability of their chemical environment (Pane and Barry, 2007).

Commercially and nutritionally important finfish appear less likely to experience direct harm from higher CO_2 levels and lower pH associated with OA, yet evidence suggests that possible food-web changes caused by OA could indirectly affect these fishes. Marine fishes with high metabolic rates and well-developed acid-base regulatory systems are believed to have sufficient capacity to respond to elevated environmental CO_2 levels (Melzner et al., 2009; Pörtner, 2008). Several studies have demonstrated that growth rates of juveniles and sub-adults of temperate and boreal marine fishes are not negatively impacted by CO_2 levels in excess of those predicted to result from OA (Foss et al., 2003, 2006). Juvenile walleye Pollock (*Gadus chalcogrammus*), an important Alaskan species, also demonstrated no significant negative effects from exposure to OA (Hurst et al., 2012). While experiments with eggs and larvae of walleye pollock did not show detrimental effects from rearing in low pH (Hurst et al., 2013), experiments on Atlantic cod (*Gadus morhua*) and Atlantic herring (*Clupea harengus*) have suggested that some commercially important boreal species can be negatively affected by OA (Franke and Clemmesen, 2011; Frommel et al., 2012). Potentially of larger concern for commercial fisheries are indirect effects: the reduction of productivity or changes in species composition of lower trophic levels that may happen as a result of OA, and the resulting effects on predatory finfish if their preferences are inflexible or prey is scarce (e.g. Kaplan et al., 2010). Successful recruitment of marine fishes is dependent upon the availability of sufficient prey resources that meet specific nutritional requirements (Litzow, 2006). Pteropods, which calcify the more soluble $CaCO_3$ mineral aragonite, are a prey for pelagic fish in subarctic and arctic regions (Orr et al., 2005). In Alaskan waters, pteropods are thought to be important prey for juvenile salmon (Aydin et al., 2005; Karpenko and Koval, 2012) and other

Table 1
Breakdown of the top 10 commercially important species, ecologically important species, and other economically important species in Alaska and the current state of knowledge regarding the physiological impact of ocean acidification on these organisms.

		Species (ranked by NMFS 2011 economic value)	Physiological impacts	References
Top 10 commercially important species	1	Walleye pollock, *Theragra chalcogramma*	Increase otolith deposition rate in juveniles	Hurst et al. (2012) and Hurst et al. (2013)
	2	Sockeye salmon, *Oncorhynchus nerka*	N.D.	
	3	Pacific halibut, *Hippoglossus stenolepis*	N.D.	
	4	Pacific cod, *Gadus macrocephalus*	No reduction in growth efficiency	Hurst et al. (unpublished data)
	5	Pink salmon, *Oncorhynchus gorbuscha*	N.D.; modeled growth decreases if pteropods decline	Aydin et al. (2005)
	6	Sablefish, *Anoplopoma fimbria*	N.D.	
	7	Snow Crab, *Chionoecetes* (any)	Uncompensated acidosis in Tanner crab	Pane and Barry (2007)
	8	King Crab, *Paralithodes*	Lower survival, growth, and calcium content	Sigler et al. (2008)
	9	Chum salmon, *Oncorhynchus keta*	N.D.	
	10	Yellowfin sole, *Limanda aspera*	N.D.	
Ecologically important species		Shrimp, *Pandalus borealis*	No negative effects on larval fertilization success or development time	Bechmann et al. (2011)
		Copepod, *Calanus glacialis*	No significant effect on egg production; pH 6.9 delayed egg hatching and reduced overall hatching success	Weydmann et al. (2012)
		Shelled pteropod, *Limacina helicina*	CaCO3 precipitation rate decrease, shell exterior dissolution	Orr et al. (2005), Fabry et al. (2008), Comeau et al. (2010 Plos One), and Bednaršek (2014)
		Cold water corals, *multiple*		Guinotte and Fabry (2008), Fish and crabs, particularly juveniles, use coral habitat as refuge and as focal sites of high prey abundance Stone et al. (2005)
Other economically important species		Dungeness crab, *Cancer magister*	Temporary acid-base shift followed by compensation	Pane and Barry (2007)
		Spider crab, *Hyas araneus*	Slower larval development and reduced larval growth and fitness	Walther et al. (2009)
		Edible crab, Cancer pagurus	High CO2 and temperature enhanced sensitivity, reduced protein synthesis rate	Metzger (2007)
		Pacific oyster, *Crassostrea gigas*	Decreased growth and survival	Gazeau et al. (2007) and Waldbusser (2001)
		Olympia oyster, *Ostreola conchaphila*	N.D.	
		Pinto abalone, *Haliotis kamtschatkana*	Decreased larval survival, increased shell abnormalities	Crim et al. (2011)
		Weathervane scallop, *Patinopecten caurinus*	N.D.	

harvested species (Moss et al., 2009). Recent studies of natural pteropod populations in the Southern Ocean in conditions similar to those currently observed in coastal Alaska have shown rapid and significant shell dissolution (Bednarŝek et al., 2012).

Risk assessments

The heavy dependence of humans on marine organisms in Alaska implies that ecosystem services based on these species could change as OA progresses (Cooley et al., 2009). Early studies of OA's potential human impacts have focused on direct macroeconomic losses likely from specific scenarios of dependence, commercial harvests, and damages to marine species (Brander et al., 2012; Cooley and Doney, 2009; Narita et al., 2012). However, indirect microeconomic impacts due to climate change are also likely to manifest, such as changes in food security or shifts in livelihoods (e.g. Allison et al., 2009; Battisti and Naylor, 2009; Cooley et al., 2012; Lobell et al., 2008). Because Alaskans involved in the fishery sector may have alternatives for employment, food sources, and recreational activities, risk assessment offers a more flexible approach for considering the complex landscape of factors affecting community risk.

In this study, we used the best available and most recent chemical, biological, and socio-economic data specific to Alaska to assess current patterns of human dependence on marine resources within the state that could be negatively impacted by OA. Using a risk and vulnerability framework based on the Intergovernmental Panel on Climate Change (IPCC) Special Report on Managing the Risks of Extreme Events and Disasters to Advance Climate Change Adaptation (SREX; Cardona et al., 2012), we relate multiple oceanographic variables to Alaskans' dependence on fisheries and marine ecosystem resources, while also considering demographic and nutritional characteristics of regional human communities around the state. In Section 'Materials and methods', we describe the framework, geographic regions addressed, and other data used. By synthesizing multiple datasets, we were able to make an initial assessment of current conditions throughout the state. From this we developed an overall index assessing the risk from OA for Alaska that incorporates all of these data. In Section 'Results', we present the results of this analysis as they relate to hazard, exposure, and social vulnerability, and their integration. In Section 'Discussion', we discuss the results in context with other studies, and in Section 'Conclusions', we present some brief conclusions and possible links between this work and local policies.

Materials and methods

Components of ocean acidification risk index

We have structured the assessment using the risk and vulnerability framework developed by the IPCC SREX (Cardona et al., 2012) for climate change, so that our findings can be evaluated in a common structure and language with other social–ecological risks from climate change. In this framework, we consider the overall "disaster risk" related to OA. Disaster risk is shorthand for the likelihood that extreme physical events will intersect with vulnerable social groups, resulting in negative effects that will require emergency intervention (Field et al., 2012). Although OA may not represent the same class of climate disaster as, for example, severe flooding, it could disrupt human livelihoods and nutrition over annual to decadal timescales, as it already has in the Pacific Northwest of the U.S. (Washington State Blue Ribbon Panel on Ocean Acidification, 2012). In that way, OA is more analogous to climate change's effects on agriculture, which could be both long- and short-term and act through direct organism and indirect ecosystem routes. The evaluation of the disaster risk related to OA using this framework provides broad-based insights into possible ways to offset its risk, because this approach weighs natural hazards equally against socio-economic resources and liabilities.

The framing of risk and vulnerability we applied here differs slightly from the few studies that previously evaluated risk of losses from OA (i.e. Cooley and Doney, 2009; Brander et al., 2012; Narita et al., 2012). Using Cardona et al.'s (2012) definition, we assessed risk using three main components: hazard, exposure, and vulnerability. Vulnerability is made up of two dimensions, referred to as sensitivity and capacity. Within this framing, the component of exposure is independent of vulnerability. This is a slight deviation in terminology from definitions described in previous IPCC reports (2007 and 2001), which evaluated exposure as one dimension of vulnerability. Lavell et al. (2012) provide a thorough discussion of the IPCC's change in definition and its stronger focus on risk. In brief, by separating exposure out of vulnerability, they maintain that vulnerability is a latent trait of a system (social, ecological, or other), and thus can be described as independent of the hazard. Here, vulnerability is constructed based on social systems' *sensitivity* and *adaptive capacity*, concepts that are described in Section 'Vulnerability' below.[3]

The total disaster risk is based on the intersection of the hazard, the assets exposed to the hazard, and a system's vulnerability to the hazard (Fig. 2). Here, OA is the environmental hazard (navy blue region in Fig. 2) and how it is projected to change over time. Exposure refers to where organisms that could be harmed by OA are located. For this exposure component of the analysis, we focus only on living marine resources that are directly important to human communities in Alaska. Our analysis of vulnerability of the social system depends on two components: (1) Sensitivity, here defined as human communities' degree of dependence on OA-susceptible resources, which is offset by the presence of alternative resources; and (2) adaptive capacity, here encompassing the human communities' estimated ability to respond with proactive adaptation in anticipation of an environmental challenge. Here, we assessed the overall risk from OA as a function of all of these contributing factors.

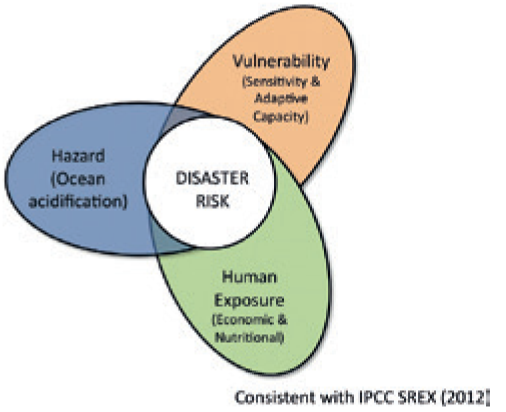

Fig. 2. Schematic of risk framework used in this study, adapted from IPCC SREX. Disaster risk is a function of the intersection of a hazard (here, OA), exposure of human communities to the hazard/disaster, and vulnerability of the human system to changes caused by these factors.

Geographic analysis units

The fisheries and socio-economic datasets used here were originally divided into specific geographic regions that were not all aligned. For example, fishing management areas are unique at the species level while fishery harvest data relates to landing ports and land-based census areas. To compensate for these geographic differences between datasets, we chose to spatially fit each distinct dataset into the standard federally assigned census areas and boroughs for the state of Alaska (Alaska Department of Labor and Workforce Development, 2011).

The marine coastal zones around Alaska can be broken down into four broad regions and extend roughly from the shoreline to the outer continental shelf, which is where nearly all commercial and subsistence harvest is conducted. These regions are the Gulf of Alaska (GOA), the East Bering Sea/Aleutian Islands (EBA), the Chukchi Sea (CS), and the Beaufort Sea (BS). Each region is impacted by unique biogeochemical processes, and we have broken them down here based on a number of factors including general ocean circulation patterns, rates of production, distribution of fisheries, and sea ice extent. The geographic extent of each region varies considerably, with the EBA and CS covering large regions due to their broad continental shelves, while the GOA and BS cover much smaller areas.

Hazard

We quantified the hazard of OA using surface pH and the saturation state of the two $CaCO_3$ minerals aragonite (Ω_{arag}) and calcite (Ω_{cal}). To illustrate the past, present, and projected chemical properties in Alaskan waters, we use output from the coupled climate-ocean model NCAR CESM1-BGC (National Center for Atmospheric Research Community Earth System Model with a biogeochemistry model enabled; Lindsay et al., 2014; Long et al., 2013a). The CESM1-BGC model is a descendant of the NCAR CCSM4 model, differing only in the inclusion of an ocean biogeochemical model (Moore et al., 2004) and three-dimensional atmospheric CO_2 tracers, which are each interactively coupled to each other and to the

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[3] There are many ways that scholars and practitioners employ the terms vulnerability and risk (differences discussed in, but not limited to, Cardona (2004), Cardona et al. (2012), Cutter et al. (2008), and Füssel (2007)). We sought to avoid adding to the confusion of the literature. For this reason, we remain consistent with conceptual framing of the most recent IPCC reports because these represent a consensus on the complicated topic among a large number of international experts that specialize in the concepts' theoretical derivations.

land biogeochemistry model (Lindsay et al., 2014). These improvements allow additional biogeochemical feedbacks to the physical climate (Lindsay et al., 2014). Historical atmospheric CO_2 emissions were used to force the simulation over the industrial period (Lindsay et al., 2014), while the future simulation followed the atmospheric CO_2 emissions from the IPCC RCP 8.5 (van Vuuren et al., 2011) scenario. CESM-modeled seawater CO_2 levels are therefore influenced primarily by atmospheric CO_2 levels, ocean physics, and respiration and primary production in the ocean. The simulated changes in surface pH, Ω_{arag} and Ω_{cal} from the past (1880–1889) to near-present (2003–2012) and end of this century (2090–2099) are calculated as decadal mean values for each region (see Section 'Geographic analysis units') and time period.

For the hazard component of the risk framework, we ranked the four ocean regions according to the forecasted level of decrease in decadal mean Ω_{arag} between the near-present (2003–2012) and the end of this century (2090–2099). We anticipate that ocean regions with greater projected decreases in mean surface Ω_{arag} will be at greater risk of the hazard represented by OA. Each of the marine geographic regions abutted multiple census areas on land, so multiple census areas experienced the same hazard. Inland census areas were assigned the mean change in surface Ω_{arag} for the four ocean regions. Relative hazard was scored by ranking the ocean regions from 1 (smallest projected $\Delta\Omega_{arag}$) to 4 (greatest projected $\Delta\Omega_{arag}$).

Exposure

Exposure to the effects of OA is related to which marine resources are important to human communities and susceptible to OA. In a vulnerability and risk analysis, exposure typically is guided by the question of "who or what will be exposed to the given hazard or stressor?" In our application of the framework, exposure refers to where organisms are located that could be harmed by OA. However, absent of the specific locations of the organisms, we represent their distribution with two metrics that indicate the organisms' relative importance to certain areas. Metrics of importance of organisms are based on two social values: economic and nutritional. The side benefit of using social importance to represent the geographic location of organisms is that it more accurately represents the aspect of the organisms we are interested in for this risk assessment.[4] Given that OA affects some marine organisms more than others (and in different ways), we accounted for this difference using the state of the science as it applies to marine organisms in Alaska. To date, only a few of the important fishery species in Alaska have been tested for a response to OA (Hurst et al., 2012, 2013; Long et al., 2013a,b), and these physiological sensitivity evaluations do not yet provide comprehensive evaluations of the full range of potential OA impacts and their consequences to population- or ecosystem-level dynamics. But it is standard to assume that negative effects on individuals will result in some degree of negative population-scale consequence (Kroeker et al., 2013a,b). Furthermore, Alaskans' direct and measurable use of OA-susceptible species is primarily through fishing and fishery-related activities, so marine biological data used in this study are restricted to species with important commercial or subsistence harvests.

To transform our knowledge of how Alaskan species are likely to respond to OA (Table 1 and Section 'Introduction') with socioeconomic effects, we sought to quantify the value of individual species to Alaskans. However, data reporting confidentiality rules and the lack of research on individual Alaskan species required that we pool harvest data into increasingly coarse biological categories as our analysis became more geographically detailed. For statewide analyses, confidentiality-based data gaps were minor at the species level, so we presented data by species or by major functional groups as in Cooley and Doney (2009). This grouping allowed for some consideration of the differential species-specific responses exhibited by Alaskan species. For census area-scale assessments, both commercial and subsistence harvest data were pooled into three major groups: shellfish, salmon, and other finfish. The shellfish category included all species of crab, shrimp, clams, octopus, squid, scallops, urchins, and sea cucumbers; the salmon category included all species of salmon; and the "other finfish" category included everything else. This grouping strategy reflected the finding that mollusks from other regions experience net negative effects from OA (Kroeker et al., 2013a,b), and that red king crabs and Tanner crabs from the Alaska region also exhibit negative responses from OA (Long et al., 2013a,b; see also Table 1). The negative responses of these taxa are due to direct effects, such as changes on calcification, growth, survival, development, and abundance (Kroeker et al., 2013a,b).

It is also expected that Alaskan species that prey on calcifying species would experience indirect negative effects associated with food web shifts or prey abundance decreases; this possibility has been proposed for pink salmon (*Oncorhynchus gorbuscha*), which prey on pteropods, a potentially vulnerable zooplankton group (Aydin et al., 2005; Fabry et al., 2008; Bednarŝek et al., 2012). Moreover, an ecosystem-based model for the California Current projected substantial declines in harvests of species that prey on calcifiers in scenarios including OA (Kaplan et al., 2010). Because of salmon's distinct life history and the importance of their harvests to Alaska, we included them as a separate biological group at risk from the indirect effects of OA. We assumed other finfish harvests were unlikely to be significantly affected by OA; this decision was based on the lack of information demonstrating direct negative responses to OA or strong trophic linkages of specific species or finfish groups to vulnerable prey such as pteropods. Mathematically, this distinction between directly affected, indirectly affected, and unaffected groups was made by weighting the shellfish group by 2, the salmon group by 1, and the other finfish group by 0 whenever metrics were calculated relating to the proportional contribution of each to overall fishery activities.

The first aspect of marine organisms we quantified represented their economic value to Alaskans. The commercial economic value of marine organisms was determined from species-specific commercial harvest quantity and price data provided by Alaska Department of Fish and Game (ADF&G; C. Tide, personal communication), which includes information about both commercial harvesting and processing. Commercial fishing harvest quantity data were reported by weight in pounds for the total annual harvest for 2011, the most recent year available. The ex-vessel value of these harvests, or the value of the harvests received by fishermen and before processing, is from the Commercial Operator's Annual Report (COAR) and available from Cathy Tide (personal communication). Harvest quantity per capita and all per capita calculations described henceforth were calculated with population data from the 2011 U.S. Census Survey. Confidentiality agreements between records of fishermen, vessels, and/or processors and the reporting agency (ADF&G) prevented disclosure of data where three or fewer companies were involved, and this created a few gaps in data obtained for species and/or census areas. The number of people involved in commercial fishing was determined primarily through licensing and permit data (ADF&G Administrative Services Division) and state employment estimates. All 2011 permit holders claiming Alaskan residency and having an Alaskan address were included in the number of people involved in commercial fishing. The number of crew associated with each of these permits was also

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[4] Other studies that focus on *ecological* risk of organisms to OA alternatively would be less interested in social values and more interested in the organism's role in ecosystem function.

included, based on ADF&G estimates using 2010 data. The number of people involved in processing includes both residents and non-residents, but all processing activities take place within the state (Alaska Department of Labor and Workforce Development, 2011).

Marine resources supplying food to Alaskans was the second aspect of marine organisms' exposure to OA that we investigated. Alaska residents' nutritional dependence on marine resources was gauged from subsistence fishery harvest quantity data from the ADF&G (J. Fall, personal communication). These data are reported as the total salmon, total shellfish, and all "other fish" in usable pounds per person, based on aggregated household survey data, post-season survey records, and permit data taken throughout the state, grouped into federally designated census areas for this study (J. Fall, personal communication). Subsistence fishing activities considered here included all personal, noncommercial fishing activities performed in state waters.

Human exposure to OA through fishery resources were semi-quantitatively assessed using a metric (E) that sums the measures of both commercial economic and nutritional importance. For notational clarity, when describing our index we used the term "economic" and the subscript E to refer to commercial harvest and processing activity. We used the term "nutritional" and the subscript N to refer to subsistence activity.

For each census area, we quantified exposure due to economic value, E_E, as:

$$E_E = 2C_{R,Sh} + C_{R,Sl},$$

where $C_{R,Sh}$ is the percent of total commercial revenue from shellfish, and $C_{R,Sl}$ is the percent of total commercial revenue from salmon. To overcome data gaps caused by reporting confidentiality rules, we first calculated the statewide average values for harvests of each species per company (pounds) and revenues from each species per company (dollars) for the "shellfish" and "salmon" categories. Then we estimated the additional harvest weight and revenue values that had not been reported by multiplying the number of companies in each census area whose data were confidential by these statewide average values, and adding the estimated additional harvest and revenue estimates to the reported totals.

Exposure in terms of nutritional importance of marine organisms, E_N, was calculated as:

$$E_N = 2S_{WT,Sh} + S_{WT,Sl}$$

where $S_{WT,Sh}$ is the percent of total subsistence harvest weight from shellfish; and $S_{WT,Sl}$ is the percent of total subsistence harvest weight from salmon. The resulting census area values for both E_E and E_N were divided into quartiles and scored from 1 to 4, with lowest exposure values receiving a score of 1 and highest a 4. Scored nutritional and economic exposure were then evenly weighted and combined to determine exposure:

$$E = 0.5E'_E + 0.5E'_N,$$

where the prime symbols indicate the quartile-classified quantities scored from 1 to 4 as described above.

Vulnerability

The severity of impacts from OA (and other environmental hazards) depends on not only the level of exposure of the system to the hazard, but also the degree of vulnerability of the system to the hazard. Vulnerability is the degree to which a system (social, natural, or otherwise intertwined) is susceptible to harm from a given hazard (Cardona et al., 2012). We use vulnerability to describe the social system because we are interested in the risk that OA presents to humans. There is also a growing literature that focuses on organisms, in which the term vulnerability is used to describe biological or ecological susceptibility of species or ecosystems (or their services) to a given hazard (e.g. see Foden et al., 2013; Williams et al., 2008). The concept of vulnerability is defined and evaluated in several different ways across (and even within) disciplines (Füssel, 2007). Here, we follow the terminology and conceptualization of vulnerability within risk as presented in Cardona et al. (2012). This largely stems from the disaster risk management community and has been united with the climate adaptation community in the Oppenheimer et al. (2014) and Cardona et al. (2012) as the way the IPCC now uses the terms. In this conceptualization, vulnerability is assessed independent from exposure to the hazard and rather seeks to understand those population characteristics that inherently increase Alaskans' propensity to suffer from OA. To gauge this social vulnerability, we evaluate the social system's sensitivity and adaptive capacity to OA. Here, the terms of sensitivity were viewed as the degree of human reliance on OA-susceptible organisms. In terms of adaptive capacity, this encompasses characteristics of the current socio-economic system that afford flexibility in the face of changing ecosystem services, which includes the ability of human systems to prepare for, respond to, or adapt to changes from OA.

Sensitivity

Sensitivity (V_S) is measured here using both commercial harvest and subsistence harvest data. In contrast to exposure, which primarily documents the extent that OA-susceptible species are present and valued in Alaska, sensitivity as quantified also includes scaling factors related to people's varying degree of reliance on the species. This scaling allows an assessment of the relative importance of this economic or nutritional dependence to individuals and the region.

Economic sensitivity, $V_{S,E}$, is evaluated for each census area. $V_{S,E,Q}$ is the estimated amount of revenue per capita from harvesting and processing OA-susceptible species, calculated as the estimated gross earnings of harvesters (data: Alaska CFEC (Commercial Fisheries Entry Commission), 2011) plus the wages from processing, divided by the estimated harvester and processor workforce. Estimated earnings of both resident and nonresident processor workers are pooled and treated together, assuming that nonresident processors are primarily spending their earnings within the state. The processor workers' earnings used here represent an upper bound for spending/local economic contributions, as the earnings are not traceable by the worker's residence. Due to this limitation, our estimates could place processor-worker spending up to 54% higher than it really is. This estimate is based on the ratio of non-resident workers to resident workers.

$V_{S,E,C}$ is estimated by multiplying the percent of the population involved in harvest and processing (calculated from CFEC and U.S. Census data) by the percent of commercial harvests, by weight, devoted to shellfish and salmon (calculated from ADF&G-based estimates for each census area developed using the data gap-closing procedure described in Section 'Exposure'). Once both $V_{S,E,Q}$ and $V_{S,E,C}$ are calculated, they are divided into quartiles, which equate to scores as described above. The index value of $V_{S,E}$ is then calculated as:

$$V_{S,E} = 0.5V'_{S,E,Q} + 0.5V'_{S,E,C},$$

where primes indicate the quartile-classified and scored quantities. Resulting $V_{S,E}$ values for each census area are then divided into quartiles and scored.

To assess sensitivity due to nutritional dependence ($V_{S,N}$), we examined the proportion of per capita subsistence harvest weight devoted to calcifiers ($V_{S,N,C}$) and salmon, and the quantity of subsistence harvests per capita ($V_{S,N,Q}$). $V_{S,N,C}$ is calculated as:

$$V_{S,N,C} = 2S_{Sh,WP} + S_{Sl,WP},$$

where $S_{Sh,WP}$ and $S_{Sl,WP}$ are the percent of per capita subsistence harvests by weight from shellfish and salmon, respectively. Subsistence harvest weight and composition are from ADF&G (J. Fall, personal communication). $V_{S,N,Q}$ is calculated as the total quantity of subsistence harvests per capita (pounds/person). Once $V_{S,N,C}$ and $V_{S,N,Q}$ are each divided into quartiles and scored, $V_{S,N}$ is calculated as:

$$V_{S,N} = 0.5V'_{S,N,Q} + 0.5V'_{S,N,C}.$$

$V_{S,N}$ values for each census area are then divided into quartiles and scored.

Adaptive capacity

The capacity to adapt, prepare for, or respond to the impacts of OA is the other component used to estimate vulnerability.[5] To examine this adaptive capacity, we created a metric that broadly assesses the resources Alaskans have even if current community stability is altered through changes in income or nutrition (Allison et al., 2009; Cooley et al., 2012). Very little research has investigated the multiple dimensions of adaptive capacity as related directly to OA (other than Cooley et al., 2012); however, a lot of scholarly work looks at the capacity of groups of people to deal with the damage from climate change, including a focus on fisheries (e.g. Allison et al., 2009; Berkes and Jolly, 2001; Cinner et al., 2009; Cinner et al., 2012; Coulthard, 2008; Hughes et al., 2012; Jepson and Colburn, 2013; Marshall et al., 2013). We gathered indicators to represent four main areas of capacity: economic stability, educational attainment, job diversity, and food accessibility. Economic stability is measured via four variables: personal income per capita, household dependence on Permanent Fund Dividend (PFD) payments, poverty, and unemployment. (The Alaska PFD is a financial dividend paid to all Alaska residents who have lived in the state a full calendar year.)

Economic stability. Economic stability-related data were from the Alaska Department of Labor and Workforce Development (2012a,b) and U.S. Census Bureau (2011). Per capita personal income (for the past 12 months in 2010 inflation-adjusted dollars) was taken from 2006–2010 U.S. Census Bureau data (U.S. Census Bureau, 2011). This is a similar measure to GDP per capita; however, it includes earnings classified as self-employment, which make up the majority of the fishing industry's income. Household dependence on the PFD was calculated by multiplying the average household size for each census area by the 2010 PFD allotment ($1281 per capita; Alaska Department of Revenue, Permanent Fund Dividend Division, 2011) and dividing by the median household income, from the 2006–2010 U.S. Census Bureau data (U.S. Census Bureau, 2011). Poverty is represented as the percent of people of all ages in poverty from 2006–2010 (U.S. Census Bureau, 2011). Unemployment rates are from the Alaska Department of Labor and Workforce Development (2012a). These data understate unemployment somewhat, as they do not account for unemployed people who have stopped actively searching for work (Alaska Department of Labor and Workforce Development, 2012a).

Education. Education is commonly used to represent people's ability to access and act on new information as one dimension of adaptive capacity. The indicator used for educational attainment in our study is the percent of people 25 years old and over that have completed high school or beyond, 2006–2010 (U.S. Census Bureau, 2011).

Job diversity. For people who rely on OA-susceptible organisms, though possibly unappealing, one adaptation option may be to seek alternative employment outside of fisheries. To represent the alternative employment options, we created a job diversity measure. To calculate this type of diversity, we used the total number of current industry types per census area from the U.S. Department of Labor (U.S. Department of Labor, Bureau of Statistics, 2012). This sums the number of employment divisions, or industry units, existing in each census area (i.e. transportation sector, food sector, health sector, etc.). Using these values, we are able to quantify the number of different job, or industry types that are reported within the specific census area with earnings. We assume these values are representative of job diversity. These data are reported annually and can be viewed as a quantitative measure of potentially available employment opportunities unrelated to fishing, because it counts all other trades but excludes self-employment, which is the usual classification for fishing activities.

Food accessibility. In cases where food is not abundantly available to a community, we consider this inaccessibility to reduce the community's capacity to cope with loss a food source (caused by OA). Food accessibility is estimated by the average annual food cost in a community, assuming that high food prices reflect long supply chains and inaccessible supplies, and they also make food less economically accessible to people in the community. Weekly food costs by community relative to Anchorage's food costs were determined as part of a long-term food cost survey (B. Luik, personal communication). Monthly averages for each census area were calculated from the survey data and multiplied by 12 months to yield annual food cost estimates. Survey data for March, June, September, and December were averaged for an annual value. Some communities were not surveyed in every period; in those cases, averages were taken from existing data points. The communities in the survey were matched to their respective census area (CA)/borough. Census areas that were not surveyed use data from the closest surveyed area. Anchorage prices are used for comparison, as the area is a large market, yet one whose prices behave differently from those in the contiguous U.S.

We combine these seven variables into an index of "adaptive capacity" (V_C) using the weighting and aggregating methods from Halpern et al. (2012), the Human Development Index (United Nations Development Programme, 2011), and the World Risk Report (Alliance Development Works et al., 2012). Datasets for each variable were scaled from their original values to a normalized range between 0 and 1.0. Finally, V_C is determined as a weighted sum of the scores for each of the index variables (I_1, I_2, ... , I_7) and their respective weights (α_1, α_2, ... , α_7).

$$V = \sum_{n=1}^{7} I_n\alpha_n$$

where I_1 = size of the economy, measured by per capita personal income; I_2 = poverty; I_3 = unemployment rate; I_4 = the PFD contribution per household; I_5 = educational attainment; I_6 = industry diversity; and I_7 = food accessibility. α_1, α_5, α_6, α_7 are 100, and α_2, α_3, α_4 are 33. V_C values were divided into quartiles and scored so that the lowest values, indicating low adaptive capacity, received a score of 4 and the highest values and adaptive capacity receive a score of 1.

Risk index

We developed an overall index to combine the different metrics developed for hazard, exposure, and vulnerability, and to allow relative evaluation of risk factors for each Alaskan census area. Once each metric was itself divided into quartiles and scored so that low

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[5] We do not differentiate between coping and adaptive capacities here (see Cardona et al., 2012), but acknowledge that this may be the topic of future social science research that seeks to contribute to the OA field.

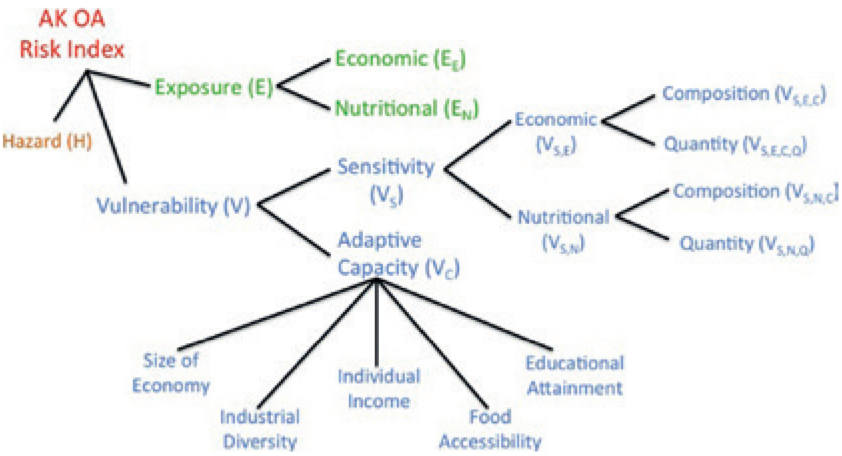

Fig. 3. Components of the risk index. Each branch is evenly weighted relative to others at the same level.

exposure, hazard, or vulnerability received low scores, etc. (indicated symbolically by primes), the final index *I* was calculated assuming even weighting of each component (see Fig. 3):

$$I = 0.33H' + 0.33(0.5E'_E + 0.5E'_N) + 0.33\Big(0.5V'_C + 0.5\Big(0.5\Big(0.5V'_{S,E,C} + 0.5V'_{S,E,Q}\Big) + 0.5\Big(0.5V'_{S,N,C} + 0.5V'_{S,N,Q}\Big)\Big)\Big)$$

$$I = 0.33H' + 0.33(0.5E'_E + 0.5E'_N) + 0.33\Big(0.5V'_C + 0.5\Big(0.5\Big(0.5V'_{S,E,C} + 0.5V'_{S,E,Q}\Big) + 0.5\Big(0.5V'_{S,N,C} + 0.5V'_{S,N,Q}\Big)\Big)\Big)$$

In the above equation, the first term relates to the hazard, the second to exposure, and the third to vulnerability. Last, we divided *I* into three levels, corresponding to lowest risk, moderate risk, and highest risk, respectively.

Results

Hazard

Model simulations (Fig. 4) indicate a rapid progression of OA in Alaskan waters, with a southward shift of habitats suitable for OA-sensitive organisms. Modeled preindustrial pH was highest in the BS region ($\mathrm{pH}_{\mathrm{preind}}$ 8.17, Fig. 4a); however, model results also indicate that this region experienced the largest preindustrial to present day change of pH ($\Delta_{\mathrm{pH}} = 0.14$), which is above the global average change of surface pH (e.g. Feely et al., 2004) and consistent with recent observations in the region (e.g. Cross et al., 2013). Present-day simulations show mean surface pH values of 8.03–8.05 in all four regions (Fig. 4b), with the largest future surface pH changes projected for the BS ($\Delta_{\mathrm{pH}} = 0.37$), where mean surface pH is forecasted to decrease to 7.66 by the end of the century (Fig. 4c). There were large regional differences in preindustrial Ω_{arag}, which was lowest in the BS (~1.4) and highest in the GOA (2.07). Since the preindustrial era, surface Ω_{arag} in the BS decreased by 0.37 units, pushing the system close to year-round aragonite undersaturation. The fastest future change in Ω_{arag} is projected for the GOA ($\Delta\Omega_{\mathrm{arag}} = 0.79$), which will lead to a shoaling of the aragonite saturation horizon by 179 m. By 2100, all waters around Alaska are projected to be perennially undersaturated with regard to aragonite, and waters in the BS and CS are even projected to be undersaturated with regard to calcite during parts of the year. Only habitats within EBA and GOA will remain supersaturated with respect to calcite during this century, with $\Omega_{\mathrm{cal}} = 1.31$ and $\Omega_{\mathrm{cal}} = 1.52$, respectively. Table 2 summarizes the changes between past, present, and future.

Exposure

Because risk to the human populations in Alaska from OA operates through potential changes in populations of specific marine species, we considered exposure to only include OA-susceptible species that are also important to humans. Although Alaska's largest fisheries, both by revenue and by weight, rely on finfish such as pollock, salmon, and halibut, a substantial portion of both commercial and subsistence fisheries rely on mollusks and crustaceans. The ADF&G has published summer harvest distribution maps for some crab and clam species, which we have merged (Fig. 5), illustrating the intersection of the oceanographic hazard of OA, the presence of OA-susceptible species, and human uses of these species. Most commercially harvested crab species are primarily taken from Bristol Bay and the Bering Sea, while commercially and nutritionally important Tanner crabs are found in the GOA near the coast. In contrast, clams popular with subsistence harvesters are located very near shore, along the Aleutians and all along Alaska's southern coast bordering the GOA. In other basins, there is evidence that warming has resulted in shifts of finfish species (Cheung et al., 2013), but the effects of OA on biogeography are still not well known.

To examine exposure of OA on Alaskan harvests in more detail, we divided the list of Alaskan commercially harvested species for 2011 reported by ADF&G into major taxonomic groupings as in Cooley and Doney (2009): clams, scallops, urchins, shrimp, crabs, calcifiers' predators, top predators, those unaffected by OA, and those whose response to OA is unknown (Fig. 6). Calcifiers' predators dominate the array of species caught. However, when commercial harvest data are viewed by weight and revenue, a more complex picture emerges (Fig. 7). Although calcifiers' predators also lead the commercial harvest by quantity and revenue, the commercial importance of crabs and top predators also emerges. Table 3 provides insight into the completeness of these data; for most taxonomic categories, we have data from the majority of companies purchasing these species and the majority of species being harvested. Only data for scallops (1 species, 2 companies) and echinoderms (1 species, 1 company) remained confidential; we assume the small number of species and companies involved

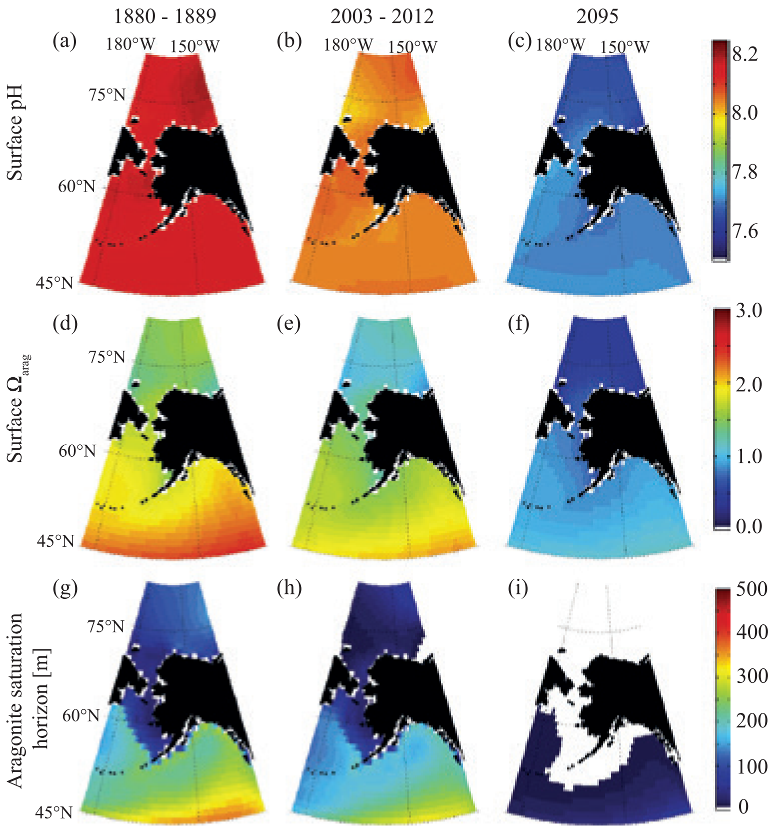

Fig. 4. Surface (top) pH, (middle) Ω_{arag}, and (bottom) saturation horizon (m) for (left) 1880–1889, (middle) 2003–2012, and (right) 2095, calculated from CESM output.

Table 2
Modeled average changes (Δ) in Ω_{arag}, Ω_{calc}, pH, temperature, and salinity from the past (1880–1889) to the present (2003–2012), and the present to the future (2095).

Parameter	Change in parameter (Δ)	Chukchi Sea	Beaufort Sea	Bering Sea	Gulf of Alaska
Ω_{arag}	Present–past	−0.30	−0.37	−0.26	−0.31
	Future–present	−0.59	−0.52	−0.68	−0.79
Ω_{calc}	Present–past	−0.48	−0.59	−0.42	−0.50
	Future–present	−0.94	−0.84	−1.09	−1.26
pH	Present–past	−0.11	−0.14	−0.10	−0.10
	Future–present	−0.35	−0.37	−0.35	−0.34
Temperature	Present–past	0.52	0.41	1.28	1.25
	Future–present	2.70	2.48	4.15	3.40
Salinity	Present–past	−0.48	−0.52	−0.12	−0.07
	Future–present	−0.94	−0.05	−0.51	−0.40

suggests that these harvests are relatively small compared to others.

Examining commercial harvest characteristics by CA shows some important regional patterns (Figs. 8 and 9). Revenues from shellfish are most important in southeast and southwest Alaska (Haines through Wrangell and Aleutians East through Lake and Peninsula, respectively). Revenues from salmon are important everywhere in the state, but especially in Alaska's interior and western CAs. When we filled gaps associated with unreported harvests or revenues using the state average per company multiplied by the number of companies not reporting, we found that we overestimated both shellfish harvest weight and revenue by 4% and underestimated salmon harvest weight and revenue by 1–2%, compared to ADF&G-reported statewide total harvest weights and revenues. We conclude that the estimates we applied to fill confidentiality-based data gaps do not materially change the results of the investigation.

Finally, exposure to OA through nutritional importance was explored using subsistence data. Salmon constitute 20–85% of subsistence diets throughout Alaska (Fig. 10) and is an especially large component of subsistence diets in the interior and western CAs. Shellfish are a large component of subsistence diets in southeast Alaska and are also important in subsistence diets in south central and southwest areas. Meanwhile, other fish contribute 20–70% to subsistence diets statewide, with a mode around 35%. Assuming proportions represent some measure of preference, salmon and shellfish are the preferred subsistence taxa, while also being likely to suffer from OA.

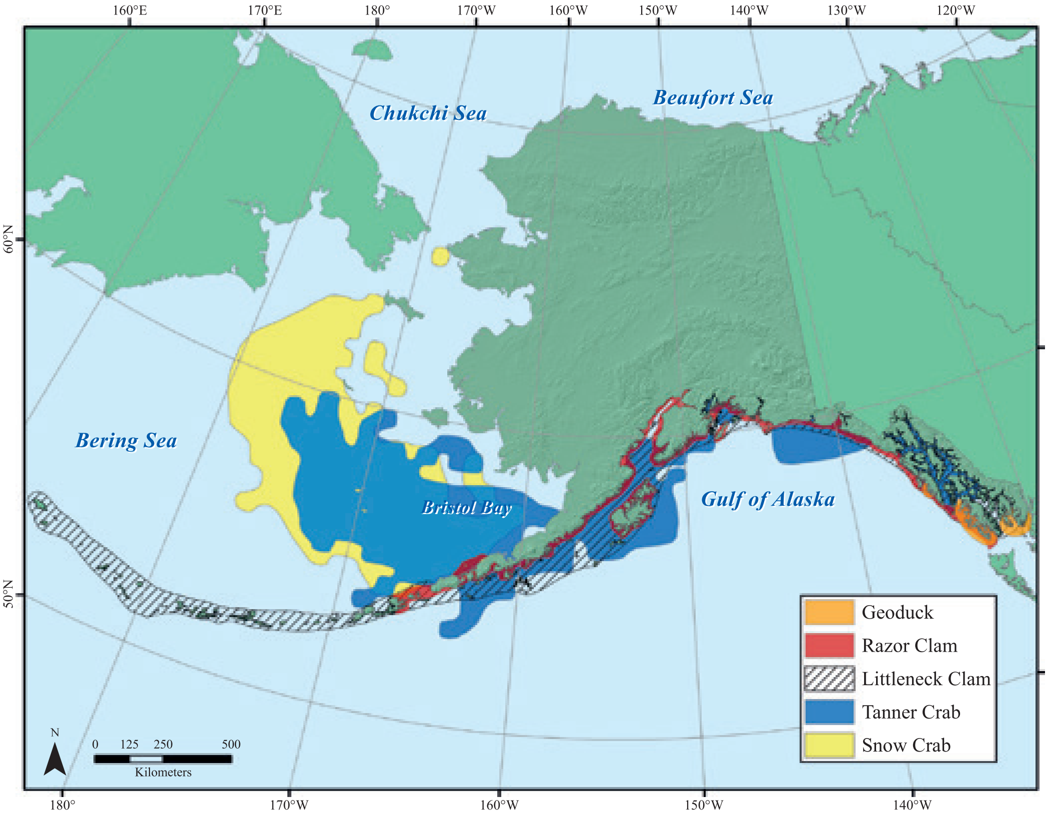

Fig. 5. Geographic range of Tanner and snow crabs, geoduck, littleneck, and razor clams (adapted from ADF&G).

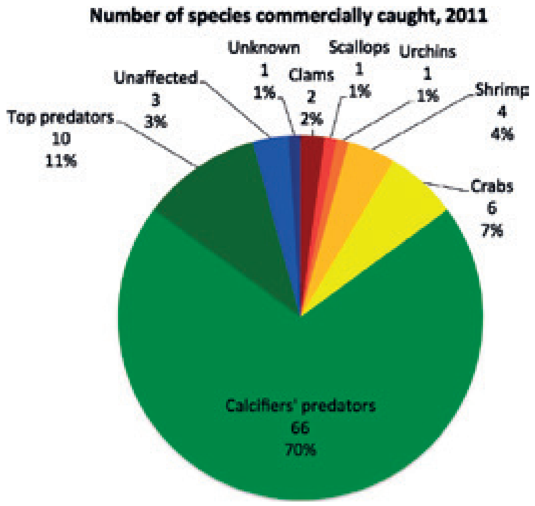

Fig. 6. The list of commercially harvested species in Alaska is dominated by finfish that prey on calcifiers during some or all of their lives or part of the seasonal cycle.

Vulnerability

Sensitivity, or the degree of economic and nutritional reliance of the human community on OA-susceptible species, makes up one half of vulnerability, as calculated in this study. Sensitivity metrics concerning nutritional dependence were based on per capita rates of nutritional dependence on OA-susceptible groups (here, shellfish and salmon). One metric was based on the proportion of OA-susceptible species consumed in a CA, and the other was based on the overall quantity of subsistence harvests per capita. Together, the two metrics provide insight into overall nutritional dependence on OA-susceptible groups. In southern and western Alaska, there were both high proportions of shellfish consumed per capita and large quantities of subsistence harvests per capita (especially southeast Alaska; Table 4). Northern and interior areas tended to have either a larger array of groups being harvested or a larger per capita consumption of subsistence harvests, but not both in the same community.

Sensitivity metrics concerning economic dependence were based on data concerning the per capita earnings of the population involved in commercial harvesting and processing and an estimate of the proportion of the population involved in harvest and processing of OA-susceptible species. Relative to other Alaskan regions, southern Alaska has the highest economic dependence on these species via commercial harvesting and processing (Table 4).

The other half of our vulnerability score is driven by the human community's adaptive capacity. In this study, the adaptive capacity metric was based on datasets providing insight into economic stability, educational attainment, job diversity, and food accessibility. Indicators measuring relatively higher in any of these four components in a given CA would potentially ensure that the residents had other options for employment and nutrition if shellfish or salmon harvests declined due to OA. In more rural areas, such as some parts of interior Alaska, northwest Alaska, and southwest Alaska, adaptive capacity was comparatively low (Table 4).

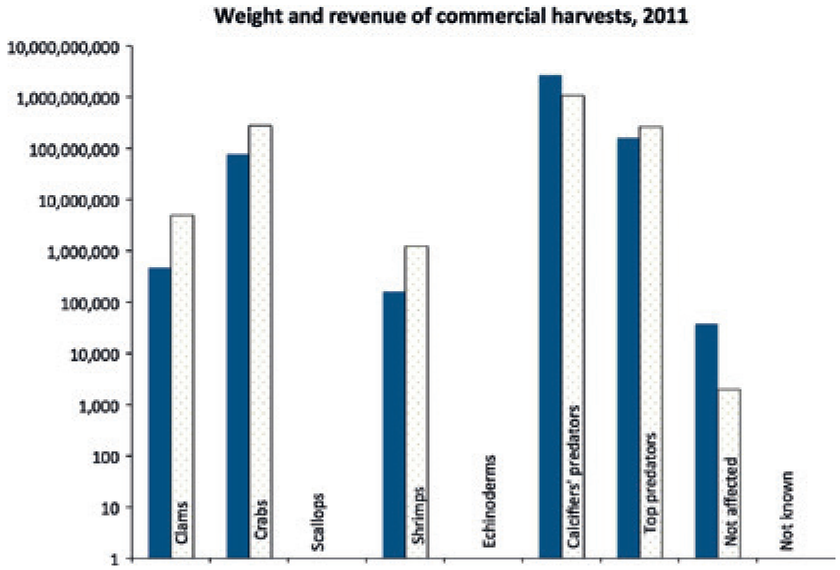

Fig. 7. Commercial harvest ex-vessel harvests (filled bars) in pounds and earnings (open bars) in U.S. dollars, by category, for 2011 based on ADF&G data. Scallops and echinoderms were unreported owing to confidentiality rules.

Table 3
For each major biological category, the proportion of species with reported data and the proportion of companies reporting data for a species in that category.

Category	Species reported (%)	Companies-species combination reporting (%)
Clams	50	85
Scallops	0	0
Urchins	0	0
Shrimp	50	93
Crabs	100	100
Calcifiers' predators	73	96
Top predators	70	98
Unaffected by OA	33	50
OA response unknown	100	100

Total risk

The total risk index brought together the risks from the hazard of OA, the extent fisheries' assets are exposed to it, and vulnerability of the human population. Table 4 shows the results for each CA, using the method described above. In general, southern Alaska is at greater risk from OA due to both dependence on OA-susceptible species for nutrition and income, and the rapidly forecasted change in chemical conditions (Fig. 4). Additional risk factors include being in a rural area with low job diversity, employment, and educational attainment, as well as high food costs.

Discussion

Hazard

Modeled present and future carbonate chemistry in the seas around Alaska represent average values modulated by global processes like atmospheric CO_2 uptake (Fig. 4), but the model's relatively coarse scale cannot simulate some of the features that greatly affect OA in Alaskan waters, such as sea ice melt, glacial discharge, river and groundwater runoff, and localized phytoplankton blooms or physical features. Indeed, local conditions observed are substantially affected in some locations by processes operating over short temporal and spatial scales that alter the carbonate system, as reviewed in Section 'Ocean acidification near Alaska'.

Biological responses

How, then, will projected changes to ocean carbon chemistry affect marine resources? Even though the western Arctic (CS and BS) may have the most rapid decline in aragonite saturation states, this area has limited direct connections to fisheries resources. However, the region is an important summer feeding ground for robust whale, walrus, and seal populations that are important to subsistence hunters along the coast as well as traditional cultural activities. On the other hand, the Bering Sea is predicted to progress more slowly toward increased OA, but it supports extremely valuable commercial and subsistence fisheries. Meanwhile, the loss of suitable habitat in a region may have unknown and cascading consequences for certain species in the future. As temperatures warm in the Bering Sea, subarctic species will likely shift northward (Cheung et al., 2013). However, water chemistry may have changed so that cooler habitats will also be characterized by markedly lower pH and Ω. In the most extreme scenario, these multiple stressors may combine to shrink or eliminate the environment appropriate for some species.

OA has been shown to have a substantial negative effect on red king and Tanner crab, particularly during the larval stages (Long et al., 2013a,b). Such declines in larval survival would likely affect overall population productivity through reduced recruitment, ultimately reducing the number of crabs available for commercial harvest. However, these early life stages occur from January to June and do not currently coincide with undersaturation events. More research on the effects of OA on other life history stages and their physiological responses is necessary to fully understand the effects it will have on crab populations throughout the year as well as other benthic calcifying organisms.

The impacts of OA on pelagic calcifying and non-calcifying organisms in the region are less clear. While there may not be a direct effect on certain pelagic finfish, such as walleye pollock, reflected in the limited impacts on growth and mortality, it is unknown how OA will affect the food supply of these fish or their

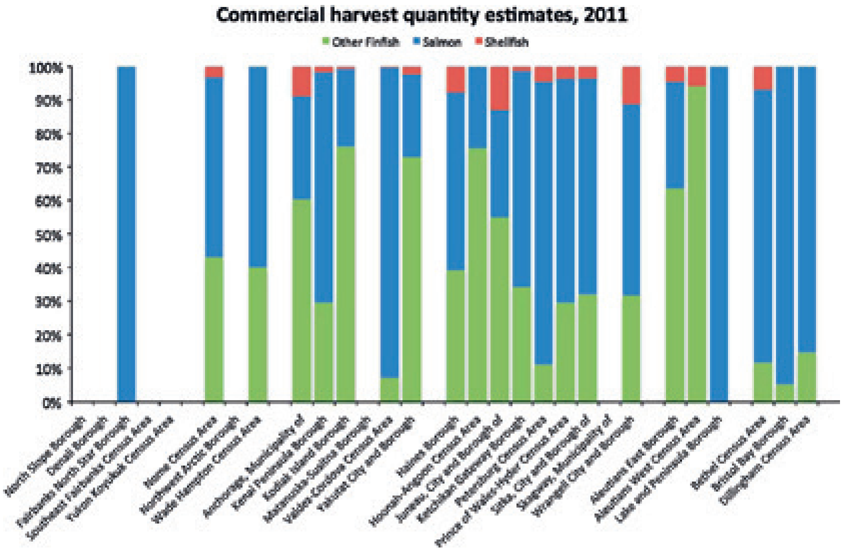

Fig. 8. Estimated proportions of 2011 commercial harvests associated with shellfish, salmon, and other finfish, calculated by weight and with confidentiality gaps estimated as described in the text. Gaps on the *x*-axis indicate no data for a given area.

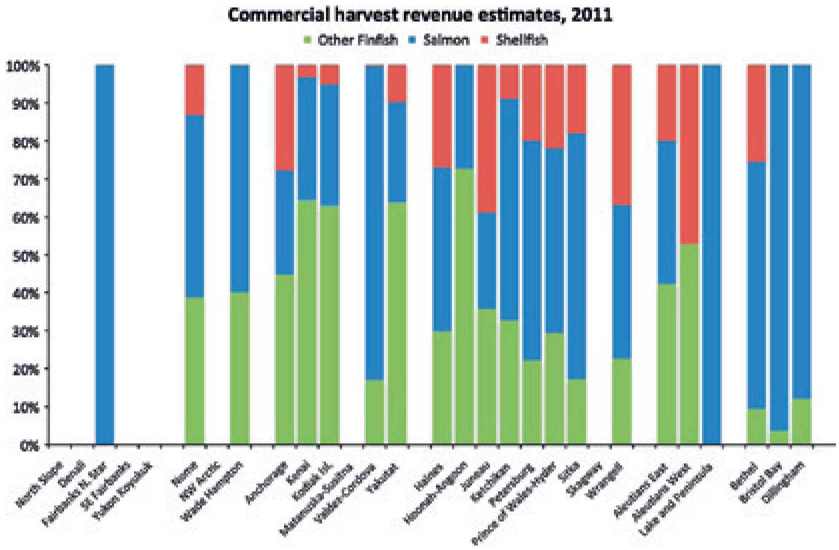

Fig. 9. Estimated proportions of 2011 commercial harvest revenues associated with shellfish, salmon, and other species, and with confidentiality gaps estimated as described in the text. Gaps on the *x*-axis indicate no data for a given area.

behavior. Walleye pollock also consume pteropods, but rely more heavily on copepods during early life stages before shifting to euphausiids as their major prey source (Brodeur et al., 2002; Dwyer et al., 1987; Moss et al., 2009). The impacts of OA on these lower trophic level organisms have yet to be resolved (e.g. Fabry et al., 2008). Because of these uncertainties and the varying degrees of organismal responses, there will likely be winners and losers as OA continues to worsen in the Bering Sea.

In addition to the potential impacts from OA-induced changes in the food web, Alaskan finfish species may also experience direct impacts from OA, as observed in some non-Alaskan species. For example, growth and survival were reduced at high CO_2 levels in newly hatched inland silversides (*Menidia berylinna*; Baumann et al., 2012); high CO_2 levels appeared to induce a range of morphological abnormalities in larval Atlantic cod without altering overall growth rate (Frommel et al., 2012); altered otolith growth (calcium carbonate ear bones; Checkley et al., 2009; Hurst et al., 2012; Munday et al., 2011) occurred in acidified conditions; and olfactory and auditory perception decreased (Dixson et al., 2010; Simpson et al., 2011). The mechanisms behind these processes or their overall effects are still not well understood, and it is possible that some Alaskan finfish species could experience these as well.

Whether indirect impacts due to trophic or habitat changes will affect Alaskan finfish is still also an open question. Copepods and krill are key links in polar food chains, and some evidence suggests that they might respond negatively to OA through changes in feeding, respiration, and excretion (e.g. Saba et al., 2012), while other studies identify mixed responses that implicate species adaptation or multigenerational acclimation (Fitzer et al., 2012). The lack of convergence of OA responses in copepods and krill prevent us from

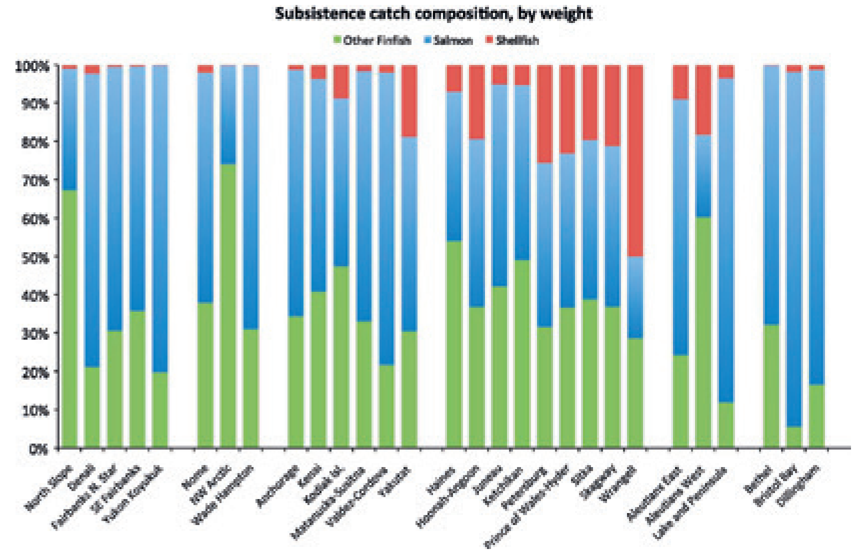

Fig. 10. Subsistence catch proportions by census area based on ADF&G data. Gaps on the *x*-axis indicate no data for a given area.

inferring any likely consequences on finfish that prey on these groups. Because of this, we treated salmon as an indirectly affected species, because evidence suggests that pteropods make up a significant and variable fraction of juvenile pink salmon diets (Armstrong et al., 2005; Aydin et al., 2005; Karpenko et al., 2007; Sturdevant et al., 2012). The projected negative effects on pteropods might represent a bottleneck in the production of this important species group. Unfortunately, while other harvested fishes are also known to consume pteropods (e.g. Moss et al., 2009; Sturdevant et al., 2012), there are insufficient data available to evaluate the general importance of this prey species to the marine fishes group as a whole. Similarly, while cold-water corals provide habitat for 42 of the 94 commercially harvested species in Alaska (J. Guinotte, personal communication), whether use of this habitat is obligatory or opportunistic is unknown, as is the degree to which changes in coral cover due to OA will translate into finfish populations. Additional research is necessary to evaluate the potential impact of OA to other marine species through these trophic and habitat pathways.

Of Alaska's many marine resource species, shellfishes appear to be the most directly influenced by OA (Long et al., 2013a,b). Declines in larval survival would likely affect overall population productivity through reduced recruitment, ultimately reducing the number of crabs available for commercial harvest. While older life stages are currently being exposed to these conditions in deeper waters and during seasonal events, the timing of reproduction in these species currently protects vulnerable larval stages from the detrimental effects of these seasonal pH minima. But the shoaling of the carbonate saturation depths and year-round persistence of undersaturated conditions will mean that crab larvae will be increasingly exposed to these conditions as OA progresses. More research on the effects of OA on other life history stages and their physiological responses is necessary to fully understand the effects it will have throughout the year on crab populations, as well as other benthic calcifying organisms.

The impacts that OA could have on Alaskan resource species through its effects on lower trophic level, pelagic calcifying and non-calcifying organisms could be more significant than the direct effects on some of those resource species. In this study, we chose to treat salmon as an indirectly affected species to explore the possibility of effects on Alaskan human communities if salmon is affected by this often-hypothesized route, rather than to make a definitive statement about OA's effects on salmon.

Human value of and dependence on OA-susceptible species

In Alaska, the principal connection between OA and the human community is via fisheries. Commercial fisheries take 97% of the wild resource harvest, while subsistence fishing produces about 2% of the total harvest of wild resource in Alaska, and sport fisheries take about 1% (Fall, 2012). Although the quantity of subsistence harvests are smaller than commercial harvests, subsistence fishing is very important to about 20% of Alaska's population, primarily Alaska Natives living in rural areas. On average, subsistence fisheries harvest provides about 230 lb of food per person annually in rural Alaska. In CAs where the average annual food cost is about twice that of Anchorage, such as Bethel, Bristol Bay, the Northwest Arctic Borough, and the Wade-Hampton CA, the nutritional (and, likely, indirect economic) benefits provided by subsistence harvests of all types of Alaskan species are immediate and critical.

The different weights applied to components of the adaptive capacity used in this study, while conservative, could be suspected to drive some of the trends observed. To judge how much the weighting affected the outcomes, we tested the effect of weighting the personal income variables in the adaptive capacity measure separately vs. weighting all seven components of adaptive capacity evenly, and we found that the effect of the weighting we applied was to slightly depress the adaptive capacity score ($R^2 = 0.97$). We also examined the effects of weighting shellfish-related numbers twice as much as salmon-related numbers. If the two taxa were weighted equally, exposure scores ($E_{H,E}$ and E_N) and vulnerability from nutritional dependence related to composition ($V_{S,N,C}$) were the only components of the final index that changed. The effects of this weighting assumption also ended up being a bit larger than the adaptive capacity assumptions, but overall, analysis results remained constant (Table 4). In that scenario, southern Alaska is still most strongly affected but the differences between census areas are more difficult to discern because there are more ties in the final index score. In the situation where shellfish and salmon are affected equally by OA, western Alaska is also more strongly at risk.

Table 4
Final scores for each census area and total ocean acidification risk index. Scores for each component of the final index (columns 3–10) are determined as described in the text. Low numbers correlate to lower risk for columns 3–11. The rank of final index indicates which region has the highest risk (#1) and which has the lowest (#29). When shellfish and salmon use are weighted equally or when the lowest future saturation state is assumed to represent the hazard (see text), the borough at highest risk changes, yet southern and western Alaska remain the regions with highest overall risk. When salmon and other finfish are weighted equally (see text), southern Alaska remains the region with highest risk.

Region	Census Area/Borough	$E_{H,E}$	E_N	$V_{S,N,C}$	$V_{S,N,Q}$	$V_{S,E,C}$	$V_{S,E,Q}$	V_C	H	Final index value	Rank of final index	Rank when shellfish/salmon equally weighted	Rank when salmon/finfish equally weighted	Rank when lowest future Ω_{ar} indicates most risk
North	North Slope Borough	1	1	1	2	2	3	4	1	1.65	29	29	27	29
Interior	Denali Borough	1	3	3	1	2	1	1	2	1.82	28	27	28	28
	Fairbanks North Star Borough	4	3	2	1	2	3	1	2	2.35	22	22	28	22
	Southeast Fairbanks Census Area	1	2	2	1	2	4	2	2	1.9	27	28	26	27
	Yukon Koyukuk Census Area	1	3	3	4	2	2	4	2	2.43	21	16	25	21
Northwest	Nome Census A Area	2	2	2	3	2	2	3	3	2.52	20	15	18	20
	Northwest Arctic Borough	1	1	1	4	2	1	4	3	2.31	24	24	23	24
	Wade Hampton Census Area	2	2	2	4	3	1	4	3	2.72	16	7	22	16
South Central	Anchorage, Municipality of	3	2	2	1	2	4	1	4	2.68	17	14	13	17
	Kenai Peninsula Borough	1	1	1	2	3	4	1	4	2.23	26	16	15	26
	Kodiak Island Borough	2	1	1	2	3	4	3	4	2.76	14	19	11	14
	Matanuska-Susitna Borough	1	2	2	1	2	4	1	4	2.35	22	16	23	22
	Valdez-Cordova Census Area	3	3	3	2	4	3	2	4	3.09	10	2	18	10
	Yakutat City and Borough	2	4	4	4	3	2	2	4	3.18	7	11	7	7
Southeast	Haines Borough	3	1	1	3	4	2	2	4	2.68	17	19	7	17
	Hoonah-Angoon Census Area	1	3	3	3	2	4	3	4	3.01	11	11	7	11
	Juneau, City and Borough of	4	1	1	1	2	3	1	4	2.81	13	22	12	13
	Ketchikan Gateway Borough	2	1	1	1	3	2	1	4	2.31	24	22	17	24
	Petersburg Census Area	4	4	4	3	4	1	2	4	3.47	5	8	6	5
	Prince of Wales-Hyder Census Area	3	4	4	3	3	3	3	4	3.51	3	13	3	3
	Sitka, City and Borough of	4	3	3	3	3	3	2	4	3.3	6	9	5	6
	Skagway, Municipality of	1	3	3	2	2	4	1	4	2.56	19	25	13	19
	Wrangell City and Borough	4	4	4	2	4	4	2	4	3.59	2	3	2	2
Southwest	Aleutians East Borough	2	4	4	2	4	2	4	4	3.51	3	3	3	3
	Aleutians West Census Area	3	1	1	2	2	2	3	4	2.76	14	19	1	14
	Lake and Peninsula Borough	4	4	4	4	4	3	4	4	3.92	1	1	7	1
West	Bethel Census Area	4	2	2	4	2	1	4	3	3.01	11	10	15	11
	Bristol Bay Borough	3	4	4	4	4	1	3	3	3.18	7	3	20	7
	Dillingham Census Area	3	3	3	3	4	1	3	3	3.18	7	3	20	7

Total picture of risk

Alaska's southern rural areas are likely at the highest risk from OA due to a confluence of factors, including: subsistence fishing for nearshore species like clams, crabs, and salmon, more rapid projected OA, lower industry diversity, economic dependence on fishery harvests, lower income, and higher food prices (Fig. 11). In particular, several areas in southeast Alaska (Wrangell City and Borough, Petersburg Census Area, and the City and Borough of Sitka) and southwest Alaska (Lake and Peninsula Borough) had scores of 3 or 4 for multiple components of the final index (Fig. 11). Even if urban areas have one or two strong risk indicators (e.g. Anchorage has risk associated with the high value of OA-vulnerable species and a faster projected change in ocean chemistry), they are offset by higher job diversity and higher overall regional income and job opportunities. This outcome is the same if adaptive capacity components are weighted evenly and if shellfish and salmon are weighted evenly. This trend, where more rural areas that have lower adaptive capacity are also more dependent on species highly susceptible to OA, matches the global trends observed by Cooley et al. (2012) for OA, Allison et al. (2009) for climate change, and Halpern et al. (2012) for ocean health and benefits overall.

Limitations of this study

Studies of human risk from global ocean changes are still at an early stage, and it is necessary to encompass a great deal of uncertainty in these studies. By focusing on OA, the path from marine biogeochemical change to human consequences seems fairly straightforward. However, there are uncertainties associated with the chemistry projections we used, the biological responses to OA, and the human community's response to changing marine harvests. Here we explore these factors and discuss them relative to the state of ocean acidification science.

Marine chemistry forecasts from a global ocean model are generally regarded as good representations of future conditions on the basin-scale, but for coastal systems, the accuracy of these forecasts is probably lower. As described in the previous sections, shallow, coastally influenced water near shore is subject to multiple pH-altering processes, and time series of pH and other carbonate system parameters in these areas are marked by spatial and temporal variability that is many orders of magnitude greater than that observed at open ocean time-series stations (e.g. Bates et al., 2012; Dore et al., 2009). This variability comes from biological production and respiration, upwelling, ice melt, and river runoff, none of which are captured in detail by the model projections used here, which show only future mean trends at the surface. Like other regions, it is very likely that Alaskan coastal marine systems already experience temporarily lower pH and saturation state as part of natural variability, and will continue to do so as the mean declines due to atmospheric CO_2. For now, the magnitude of pH and Ω variability experienced by most coastal systems is not well known until enough time-series data are gathered at nearshore locations.

In addition, the ways in which responses of individual organisms to OA translate into population- and ecosystem-scale responses are still unclear. We have only an initial but growing understanding of which aspects of OA are important for marine organisms: change in pH, carbonate ion level, saturation horizon depth, $p\mathrm{CO}_2$, seasonal variability, phenology mismatch with seasonal life cycle cues or predator–prey interactions, biogeographic shifts away from appropriate benthic habitat, and so on. We assumed that negative responses at the individual level would translate to some degree of negative response in productivity at the population scale. Certainly this has been demonstrated with the study of bivalve mollusks, which originally were reported to calcify more slowly based on studies of two species (Gazeau et al., 2007; Kurihara et al., 2007), and which are now understood to display a range of negative population-scale responses to OA, including decreased reproduction, delayed development, and lower survival, as well as lower calcification (Kroeker et al., 2013b), and ecosystem-scale shifts away from calcifying species (Hall-Spencer et al., 2008; Kroeker et al., 2013a; Wootton et al., 2008). Meta-analyses support the inference that individual Alaskan mollusks are more likely than not to suffer from OA (Kroeker et al., 2013b), but whether whole populations of harvested clams and scallops are likely to decline as a result of OA is still not well known. Likewise, the population-scale or lifetime consequences of OA on red king crab and Tanner crab are still not known, despite the recent reports of individual effects (Long et al., 2013a,b). It is entirely possible that commercially or nutritionally important Alaskan species may be able to tolerate or adapt to lower pH, lower carbonate ion conditions, and this analysis does not account for that possibility. A different approach using individual-based models including their physiological responses would probably be necessary to do so. As more experimental data become available we will also likely understand which is more important for high-latitude species: the absolute or the relative change in pH and Ω. We will also be able to better estimate the consequences of indirect trophic and habitat effects, such as the salmon-pteropod scenario we explored here and the association of many Alaskan finfish with deep-water corals that we did not explore. Additional studies may also provide insight on whether market characteristics of harvests, like meat weight, appearance, and time to harvest, will change owing to OA.

To quantify risk based on current human use patterns of marine resources also assumes that humans' use of these resources is fixed, which it is not. But by incorporating multiple indicators of exposure (economic, nutritional) and of vulnerability (adaptive capacity, degree of dependence), which reflect more systemic aspects of the social–ecological system, we attempt to provide a snapshot of risk that has enough detail to encourage a harder look at the most important factors in subsequent studies. We also experienced a challenge in aligning the boundaries of socio-economic data with fisheries/ecosystem data, and then with oceanographic data: relevant datasets are collected using different geographic divisions, units (per capita vs. percent vs. total), and on different timescales (annually, every few years, decadally). Analysis of another vulnerability index indicated that scale affected the results of the overall index (Schmidtlein et al., 2008). We attempted to close gaps using average values where possible, and used the most updated information available in every case.

The indicators of adaptive capacity used here also have significant uncertainty associated with them. First, per capita income may be highly endogenous because it is built on fish income. Second, job diversity and total regional income in many areas may be endangered from declining oil production or bolstered by new events like natural gas exports. Because Alaska's economy is an energy export economy, climate change could affect the entire economy via climate policy: a carbon tax would depress the value of exported oil. (Conversely, limiting CO_2 emissions via a carbon tax or other means could decrease OA and its effect on the marine resources being studied.) All of these factors add uncertainty to any measurement of adaptive capacity.

Building a more resilient social–ecological system in Alaska

Alaskan commercial fisheries have a long history of opportunistically switching to different species based on availability and marketability, suggesting the socio-economic system may have some ability to adapt to future conditions. An example of this is the

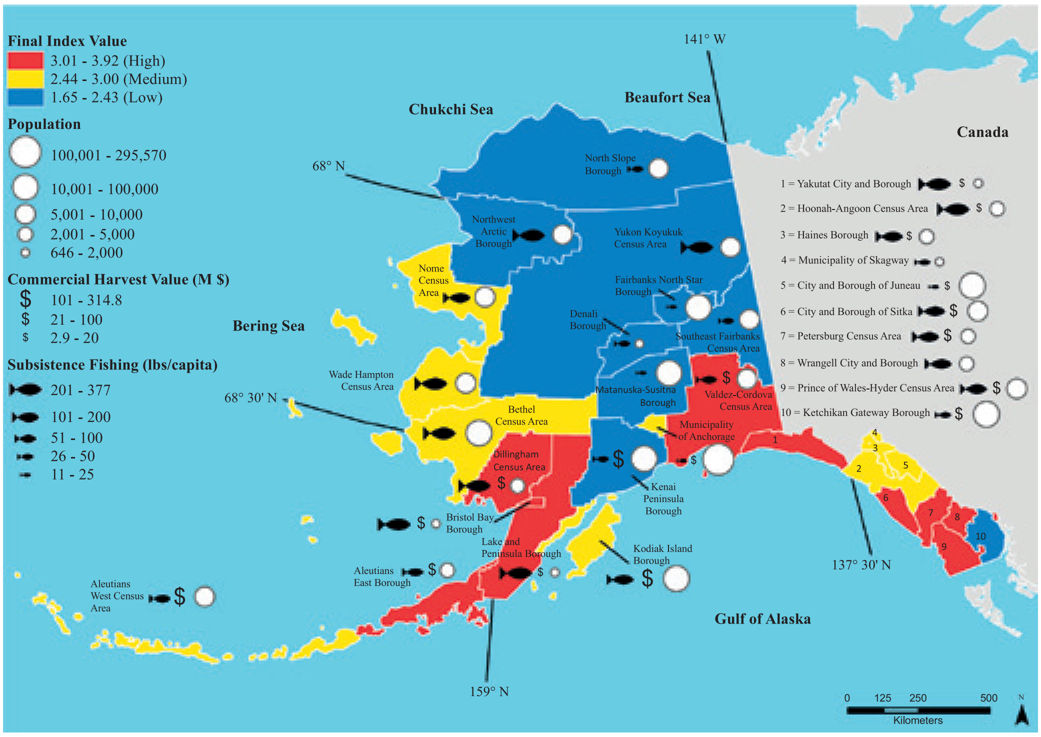

Fig. 11. Individual components of the final ocean acidification risk index for each census area.

shrimp fishery, where shrimping in the Gulf of Alaska was a major commercial and subsistence resource until the 1980s when an ecosystem change (e.g. Anderson and Piatt, 1999) caused the fishery to completely crash. Communities throughout the region were able to shift to ground/finfish fishing rather quickly.

A far more ominous example, though, is what occurred in Prince William Sound, where herring supported a major fishery until the early 1990s. It has been suggested that the 1989 Exxon Valdez oil spill initiated a decline in herring abundance (Thorne and Thomas, 2008). However, other analyses suggest that the population collapse in 1993, four years after the spill, was triggered when poor nutritional condition of herring, brought about by low zooplankton production, increased the susceptibility of herring to disease, possibly exacerbated by the stress of low winter temperatures (Pearson et al., 1999). Regardless of the underlying cause, the herring population has yet to recover and the fishery in Prince William Sound has been closed for 17 of the last 23 years (Pearson et al., 2012). The loss of this fishery cost the region millions of dollars, thousands of jobs, and the loss of a reliable subsistence food source. Many communities, especially those that were heavily dependent on the herring fishery, went into sharp decline and some small towns and village shut down completely.

The example of the region's herring loss, taken together with the community index data presented here (Table 4; Fig. 11), clearly demonstrates the need for commercial diversification, particularly in southeast and southwest Alaska. This may involve looking to other sectors, like exploiting other resources (e.g. fur seals, gold, timber, oil, and natural gas) or encouraging other industries. Many towns and villages throughout the state are facing multiple challenges, including rising food and energy costs, loss of revenue from declines in oil and gas production, and declining populations. OA may be yet another challenge to these communities, adding another stressor to a region already at socio-economic risk.

Conclusions

The outcomes from this risk assessment concerning ocean acidification's potential to affect Alaska's fishery sector can inform policies and guide future scientific studies of the social–ecological system that depends on marine resources. While we recognize that the index developed here is an intermediate step toward our full understanding of the economic and societal consequences of OA, it does provide valuable insights on social vulnerability. Community- or state-scale policies offer numerous opportunities to combat regional processes (e.g. fertilizer runoff, atmospheric emissions of nitrogen species) that worsen acidification, but only when the contributing factors are well understood. In this study, we sought to identify areas of the social–ecological system that are most vulnerable (i.e. the components of the final index with highest scores); the entire risk of the system could be decreased by the application of localized policies designed to build adaptive capacity, decrease exposure, or distribute risk where those factors were dominating the degree of vulnerability. From this analysis, it is evident that risk assessments offer more thorough decision-relevant information because they provide insight into the interaction of social, economic, and natural components instead of just one facet.

In Alaska, plans to confront OA can be made that address the natural and the social system by dealing with aspects of the hazard, exposure, and community sensitivity. To address the hazard, which is primarily associated with the natural environment, continued monitoring of conditions in Alaska's nearshore regions is an important response. This will add important regional specifics to the general picture currently available from ocean model projections, and identify which oceanographic processes are most important in driving regional OA. To address exposure to the hazard, productive responses would be most effective in southern and western Alaska, where human dependence on OA-susceptible marine resources is highest (Fig. 11). These responses might encourage diversification of fisheries activities, exploiting a broader range of species, not just those that are OA-susceptible. Ultimately, the strategies to mitigate and adapt/prepare for OA impacts must be developed by the communities themselves. To foster participation in such planning, residents and other stakeholders in vulnerable communities must first be educated about this emerging environmental challenge, then be permitted to develop response strategies that incorporate community values and are context- and situation appropriate. To reduce community vulnerability, factors that lower adaptive capacity, such as low income, nutritional status, educational attainment, or industry diversity, must be addressed. These factors create vulnerability to many environmental and social problems beyond OA, so addressing them could provide overall benefits. In Alaska, where dependence on marine resources is strong, traditional, and very deep-rooted, attempting to reduce risk from OA or any other marine-related type of global change simply by decreasing dependence on marine resources may be a poor fit. Instead, users and decision makers must consider the elements that contribute to risk, as well as those that offset it, and attempt to choose a path that optimizes these yet retains traditional and contemporary uses of these valuable marine resources.

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Annex 6

Changing ocean impacts on the key forage fish species Arctic cod in the Western Canadian Arctic – Linking climate model projections to subsistence fisheries

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Contents

A6.1 Introduction

This annex highlights the results of a study focusing on the potential impacts of ocean acidification and other climate-related stressors on marine species relevant for subsistence fisheries in the Western Arctic Bioregion. The study uses a knowledge co-production approach developed in the form of a multi-step process based on a combination of modelling and analysis tools including the Scientific Method and Indigenous Traditional Knowledge (Figure A6.1). Once all steps have been completed, uncertainties can be estimated and improvements can be made either with respect to the individual steps or to the linkages between them. The process can then be repeated, including those improvements to provide a revised assessment with reduced uncertainty ranges. The steps can be summarized as follows: (1) analyze past observed trends; (2) perform projection simulations with global and regional climate models, allowing trend estimates on 20–50 year timescales; (3) assess physiological responses and thresholds in marine species via literature research, Indigenous Traditional Knowledge, observations and focused laboratory experiments; (4) add trends, climate model projections and physiological response data to species distribution / habitat suitability and higher trophic level Ecosim/Ecopath (see Section A6.3) models; (5) assess socio-economic impacts by applying bio-economic models, evaluating current fishery-economic activities, and discussion with communities/community representatives; and (6) review law and governance. The latter addresses adaptation measures on global, regional and national scales.

This annex describes the first application of the multi-step framework in the Western Arctic Bioregion. At this point in time all the required tools have been developed, but not all components have been adequately linked. For example, while higher resolution model projections are available for the area the habitat suitability and economic models are still driven by global climate models, the Ecopath model (see Section on *The Beaufort Sea food web model*) has not yet been run into the future and physiological responses are reflected in the higher trophic level models to a limited extent, if at all. In addition, while collaboration with local communities has been established (see Section on *Community interests*), Indigenous Traditional Knowledge has been included to a very limited extent. To summarize, this case study provides an assessment that includes all required tools, but limited linkages. It has a strong focus on uncertainty analyses and the identification of gaps in knowledge. Particular emphasis is given to the key forage fish species Arctic cod (*Boreogadus saida*), since climate model projections can be linked more directly to key forage species than to the (mostly) higher trophic level species harvested.

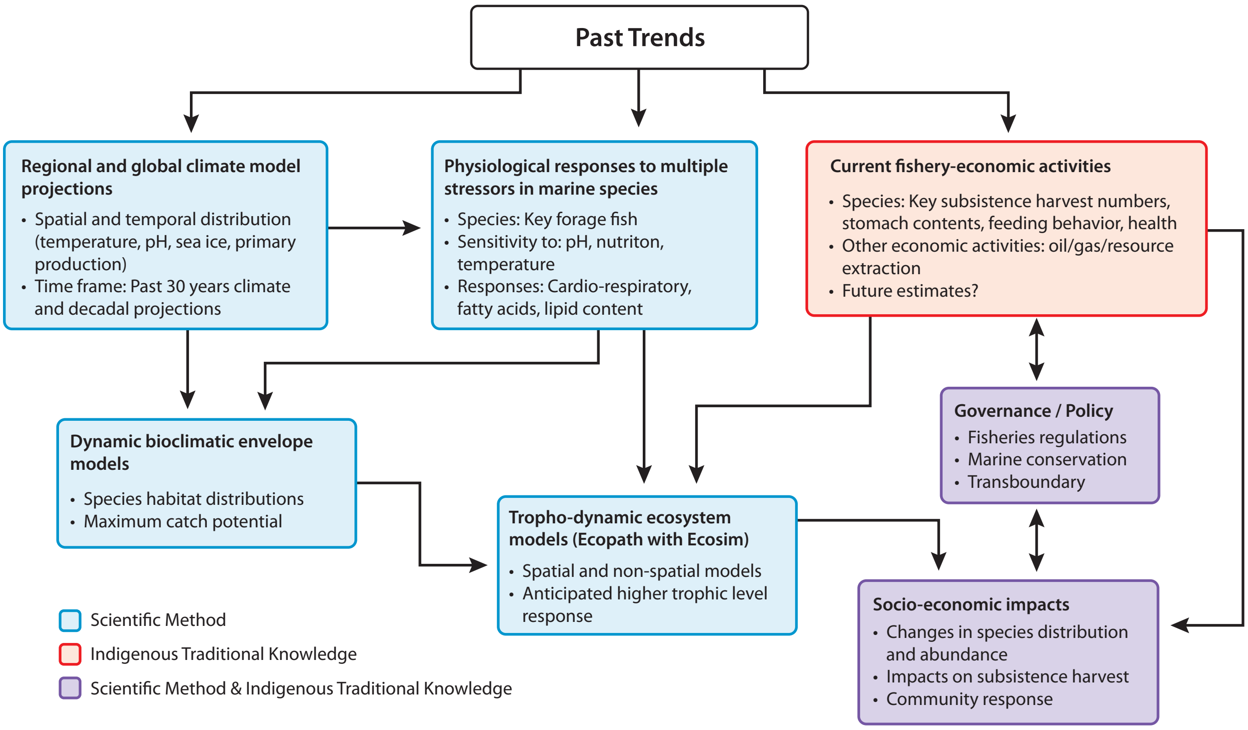

Figure A6.1 Multi-step framework linking observed trends and climate projections with impacts on subsistence fisheries. The goal is a knowledge co-production approach linking Indigenous Traditional Knowledge with the Scientific Method.

A6.2 Background

Community interests

Inuit are able to maintain a link to their traditional way of life through their harvesting practices. Hunting, fishing and trapping are both traditional and economic activities. The foods harvested from the land and sea form an integral dimension of their diet and the harvest byproducts support thriving economies based on arts and cultural practices. Hence, potential changes to subsistence harvest species are highly relevant to them. The assessment presented here is the first stage of an iterative process that has been presented to and received support from the Inuvialuit Game Council and Inuvialuit Regional Corporation. A continued collaborative effort to assess the impacts of climate and other stressors on marine ecosystems in the Western Arctic Bioregion is now linked to the Beaufort Sea Regional Strategic Assessment and includes community support for physiological studies on forage species local to the region. This effort also includes an indigenous knowledge workshop and report to be co-produced with Oceans Canada and Fisheries and Oceans Canada research teams and the Inuvialuit, and aims to synthesize comprehensive indigenous and western knowledge of forage species, links to higher trophic levels and observed changes in a rapidly changing Arctic.

Introduction to the region

The Western Arctic Bioregion encompasses the Inuvialuit Settlement Region and the Kitikmeot Region (see Figure A6.2). The associated ocean regions include the Beaufort Sea, comprising shallow shelves and Canada Basin (up to 3600 m deep), and the western and central part of the Canadian Polar Shelf, a vast shallow area with many islands. The circulation pattern on the Canadian Polar Shelf is complicated, with flows entering and leaving both bounding reservoirs from the opposite direction. The net effect is a movement of sea water from the Arctic to Baffin Bay of about 1.5 Sv/y (1 Sverdrup = 1 × 10^6 m^3/s) (Melling et al., 2008). Steiner et al. (2015) reviewed the oceanography of these areas and discussed recent environmental changes and future projections. The Inuvialuit Settlement Region was established in 1984 under the Inuvialuit Final Agreement by the Government of Canada and the Inuvialuit. It includes the north slope of the Yukon and northern Northwest Territories, the inland delta communities of Inuvik and Aklavik on the mainland, Tuktoyaktuk and Paulatuk along the mainland coast, and Ulukhaktok on the northwestern coast of Victoria Island and Sachs Harbour on the western coast of Banks Island.

The Kitikmeot Region is an administrative region of Nunavut which encompasses the southern and eastern parts of Victoria Island with the adjacent part of the mainland stretching as far as the Boothia Peninsula, King William Island and the southern part of Prince of Wales Island. Communities include the hamlets Cambridge Bay (Iqaluktuuttiaq), Gjoa Haven (Uqsuqtuuq), Kugaaruk (Pelly Bay), Kugluktuk (Coppermine), Taloyoak (Spence Bay), as well as Bathurst Inlet (Qingaut), and Umingmaktok (Bay Chimo).

Two Canadian Science Advisory Processes identified 22 Ecologically and Biologically Significant Areas (EBSAs) (DFO, 2011, 2014) in the region. In a follow-up process, ecological units (eco-units) were proposed based on a biogeographic classification system (DFO, 2015) that divided

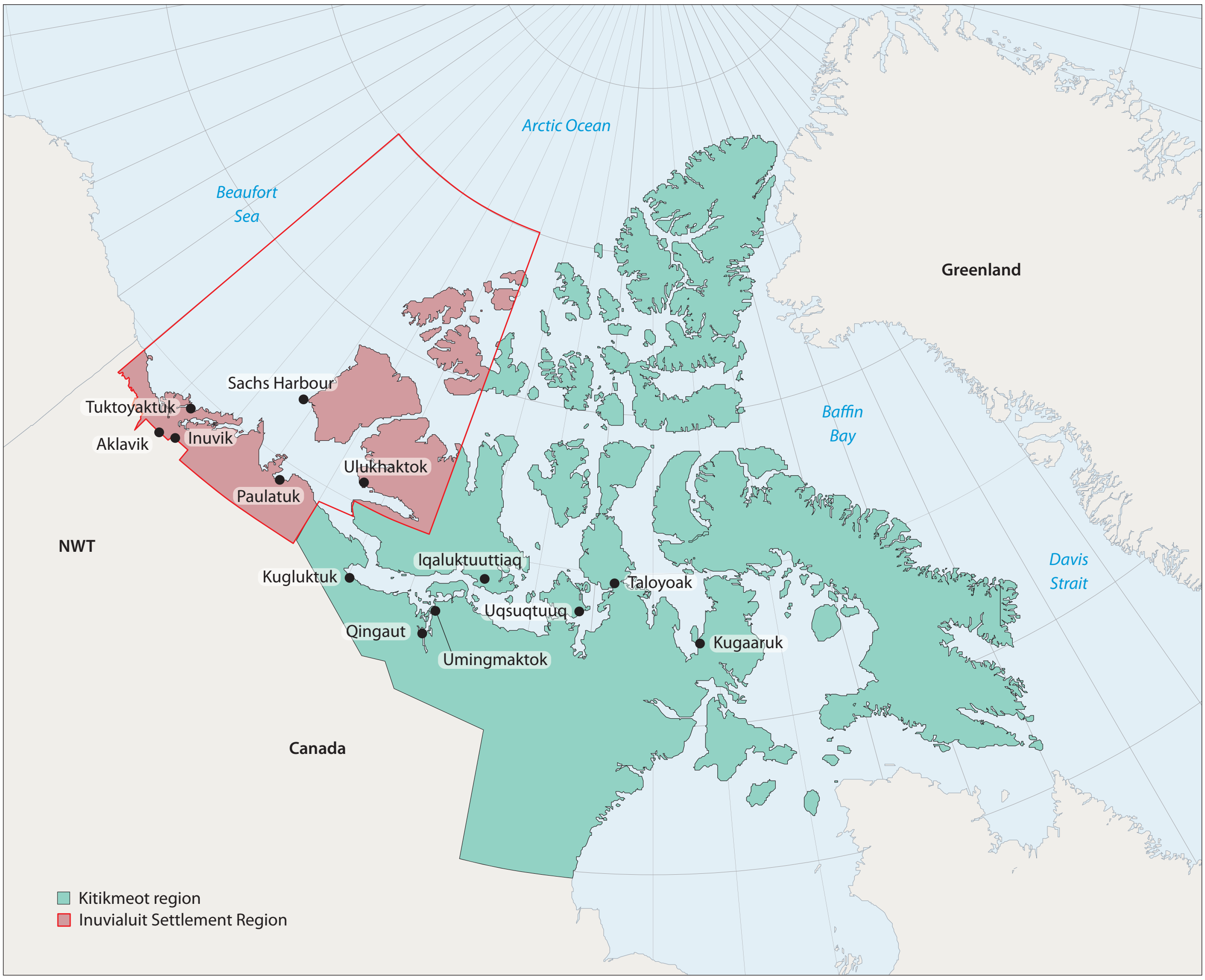

Figure A6.2 The Inuvialuit Settlement Region and Kitikmeot Region (and major communities).

the Western Arctic Bioregion into 18 eco-units (Figure A6.3). Dominant ecosystem features, sea-ice data, bathymetric data, sills and water mass information were the primary inputs used to delineate the eco-units. Proposed priority conservation areas were identified maximizing areas of overlap between the eco-units and the EBSAs to capture areas representing all habitat or ecosystem types. In total, 23 proposed priority conservation areas were identified. However, the DFO (2015) report highlighted several uncertainties: division of the bioregion into eco-units was limited by the available data and potential biases towards socially, culturally and economically important species and areas; research was focused on coastal areas in close proximity to established communities; and some of the ecosystem features, particularly those related to ice, vary spatially, seasonally and annually. Uncertainty identification is an important step within a network planning process for marine protected areas (MPAs) in the Canadian Arctic.

The Western Arctic Bioregion currently includes the recently established Anguniaqvia niqiqyuam MPA, the Tarium Niryutait MPA, and several bird sanctuaries: The Anderson River Delta Migratory Bird Sanctuary, Banks Island No. 1 Bird Sanctuary, Cape Perry Migratory Bird Sanctuary, Kendall Island Migratory Bird Sanctuary, and Queen Maud Gulf Migratory Bird Sanctuary.

Linking environmental change with subsistence harvest, health and economy

According to the recent AMAP report on climate change-related adaptation actions in the Bering, Chukchi and Beaufort Sea region (AMAP, 2017), scientific observations and traditional knowledge suggest the region is moving towards conditions never before witnessed. Environmental drivers related to seasonal loss of sea ice and increasing seawater temperatures have already led to changes in the distribution of ice-dependent or ice-associated marine mammals and birds and have initiated changes in migratory pathways and phenologies, causing changes in species composition and timing of harvests. Some coastal communities are impacted as subsistence species decline in availability or move beyond the geographic reach of harvesters. Access complications are also linked to changes in weather, storms and sea state (e.g., Steiner et al., 2015) which have made the sea rougher, less predictable and more dangerous for small boat travel, according to local Inuit (Clarke et al., 2015; Gaden and Stern et al., 2015). AMAP (2017) suggested that restructuring of harvesting strategies in response to changes in species composition and the availability of subsistence food resources may be inevitable (see also Ford, 2009; Ford and Pearce, 2010; Ford et al., 2014) and listed multiple human health issues, such as obesity, diabetes and heart disease, which have already impacted coastal communities due to the loss of a subsistence

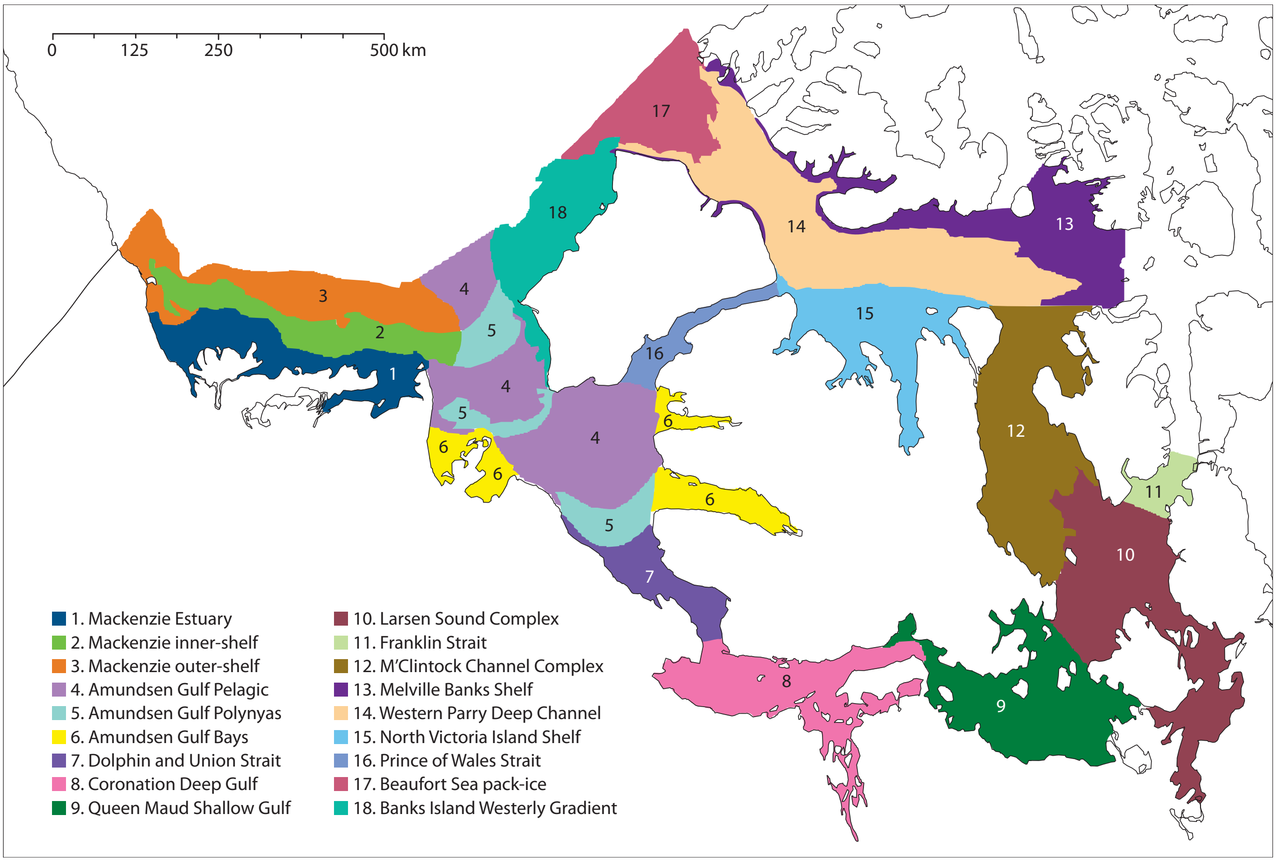

Figure A6.3 Proposed eco-units outlined in the Canadian science advisory report (DFO, 2015).

lifestyle (see also Inuit Health Survey, e.g., Galloway et al., 2010; Zhou et al., 2011; Saudny et al., 2012; Zienczuk and Egeland, 2012). The direct transmission of contaminants to human populations via subsistence foods is another impact of human drivers on Arctic coastal communities.

Information on the magnitude of seafood (and other country food) consumption by Arctic communities is critical for evaluating public health trade-offs between traditional subsistence harvest and imported (often processed) foods (Kuhnlein and Receveur, 2007). The subsistence catch of marine species is particularly important in this context, because levels of methylmercury and persistent organic pollutants tend to be highest in marine mammals and high trophic level fish species (Kirk et al., 2012; Laird et al., 2013). Although there is a general lack of information regarding subsistence fisheries, there are some studies for communities in the Canadian and US Arctic region. Little is known about patterns of traditional food use in such communities (Berkes, 1990), but it is clear that their seafood consumption is markedly higher than for the general population. For example, a meta-analysis of population and harvest data in the Canadian Arctic suggests some 70,000 people in 60 communities eat around 2600 tonnes of seafood per year (Cisneros-Montemayor et al., 2016). While such studies help provide a sense of magnitude for coastal Indigenous seafood consumption, they also highlight the need to combine local research and data with interdisciplinary modeling efforts in order to better inform climate and policy scenarios.

To understand ongoing and potential future impacts of a changing climate on Arctic residents and communities, it is necessary to recognize the close and inseparable linkage between subsistence living and the physical, economic, and socio-cultural well-being of the residents and their communities. AMAP (2017) highlighted that the traditional subsistence economy is now intertwined with and a component of a truly mixed subsistence-cash economy, with cash being required for vessels, vehicles, fuel, weapons, clothing, communication devices, and safety gear (Poppel, 2006). Hence, residents are challenged by rising fuel prices as well as by more challenging travel conditions. In the western Canadian Arctic, food insecurity is widespread and apparently increasing, while at the same time the quality of people's diet is deteriorating (Huet et al., 2012; Douglas and Chan, 2015; Watts et al., 2017). Changing eating habits are suggested to have led to enhanced risk for cardiovascular disease among a significant proportion of the population, and to deficiencies in Vitamins C and D, iron and other essential micronutrients (Egeland et al., 2011).

The decline in sea-ice cover is driving discussion about the potential for new shipping routes. The Northwest Passage along the Beaufort Sea coast, however, remains challenging because the Beaufort Gyre keeps sea ice pushed against the Canadian coast and newly open, ice-free waters are mostly uncharted, which represents a major block to further development of shipping in the area (Clarke et al., 2015). Resource extraction remains a key economic driver; the discovery of oil and gas in the Inuvialuit Settlement Region and mineral deposits in the Kitikmeot Region have had many positive benefits for communities, including improved transportation infrastructure, better employment opportunities, and the resolution of land claims (Dana et al., 2008; Lockhart et al., 2015). However, the Inuvialuit also say that they favor continued activity only if the benefits filter to them too as opposed to the oil companies and migrant employees only. They also expressed concern for the environment and about the threat that development brings to the wildlife upon which people rely as a food source (Dana et al., 2008). This shows that positive impacts on communities largely depend on local policies (such as impact benefit agreements, land claims) and the other social, cultural and environmental impacts of the development (such as those beyond financial benefits) (Knotsch and Warda, 2009).

Arctic cod – a key Arctic forage species

Boreogadus saida (Figure A6.4) is referred to as both Arctic cod and polar cod within the literature. This text uses Arctic cod exclusively. Arctic cod survive in ice-covered, sub-zero waters due to the presence of anti-freeze glycoproteins, specialized kidney function (Osuga and Feeney, 1978; Christiansen et al., 1996) and the ability to digest food at water temperatures down to -1.4°C (Hop and Tonn, 1998). A recent assessment of fish assemblages on the Beaufort Shelf and Slope (Beaufort Sea Marine Fishes Project) found that Arctic cod typifies each of the four species assemblages (near-shore shelf, off-shore shelf, upper slope, lower slope) to varying degrees. The high abundance of Arctic cod at upper-slope stations was a major structuring feature of the fish community (Majewski et al., 2013, 2015, 2017). Other studies in the Beaufort Sea have reported Arctic cod as the most abundant demersal species across a range of habitats (Rand and Logerwell, 2010; Majewski et al., 2013; Walkusz et al., 2013). Young Arctic cod (1–2 years) are often associated with the underside of sea ice feeding on ice-associated (sympagic) amphipods, indicating the importance of sympagic fauna in young Arctic cod diets (Kohlbach et al., 2016, 2017). Fatty acid profiles of Arctic cod show diatoms are the primary carbon source, indirectly obtained via amphipods and copepods (Kohlbach et al., 2017). This highlights a strong dependency of Arctic cod on ice-algae produced carbon but also suggests a particular vulnerability to changes in the distribution and structure of sea-ice habitats. Arctic cod is considered an ecological key species because it reaches high stock biomasses and constitutes an important carbon source for seabirds and marine mammals in High Arctic ecosystems (Bradstreet et al., 1986; Coad and Reist, 2004; Kohlbach et al., 2017). Figure A6.5 illustrates the primary role of Arctic cod in the transfer of energy from zooplankton to Arctic megafauna, many of which are key subsistence species harvested by western Arctic communities. Arctic cod predation moves a large proportion of the energy and nutrients from Arctic algae and invertebrates to higher trophic levels (Bain and Sekerak, 1978; Welch et al., 1993; Crawford and Jorgenson, 1996; Christiansen et al., 2012: Hop and Gjøsæter, 2013). Due to the ecological key role of Arctic cod, changes at the base of the sea ice-associated food web are likely to affect the higher trophic levels of High-Arctic ecosystems (Kohlbach et al., 2017) and hence the communities harvesting those species.

Physiological impacts

The weakest link in determining ecosystem risks and fisheries impacts of climate change and ocean acidification is the lack of information on physiological limits of key Arctic marine species to multiple stressors. This gap was addressed by undertaking a literature search on species responses to acidification (see Chapter 3) and the development of a database for Arctic and subarctic marine species (Steiner et al., 2018). The database holds information on species thresholds, lethal limits and acclimation potential for several stressors as well as knowledge gaps, where focused testing might be required. Table A6.1 presents an extract of the summary table for selected Arctic zooplankton and forage fish species as well as Arctic char (*Salvelinus alpinus*).


H. Drost


H. Drost

Figure A6.4 A young girl from Cambridge Bay helps collect Arctic cod for scientific research and Arctic cod swarm near Cambridge Bay in 2012.

The impact of global warming on the physiology of ectotherm marine species (e.g., fish, copepods, bivalves) is of particular concern due to their total reliance on habitat temperature to maintain body heat. Temperature drives metabolism and hence basic performance, including development, growth, reproduction and survival (Brown et al., 2004). However, the physiology and ecology of fish can also be affected by water pH (e.g., acid excretion across gills is dependent upon water pH).

The Aquatic Species Physiological Limits Database (ASPL Database) (Steiner et al., 2018) includes data for cardio-respiratory performance of ectotherm species and this is a powerful tool for quantifying the impact of environmental stress on animal health (Farrell et al., 2008; Anttila et al., 2013; Ferreira et al., 2014; Drost et al., 2016; Farrell, 2016). Absolute aerobic scope defines the capacity to deliver additional oxygen to support specific activities beyond basic needs (Fry, 1947; Farrell, 2016). Oxygen demand for activities such as growth, reproduction, locomotion, immune competence, and predator avoidance can influence species distribution and ultimately survival (Pörtner and Knust, 2007; Pörtner and Farrell, 2008;

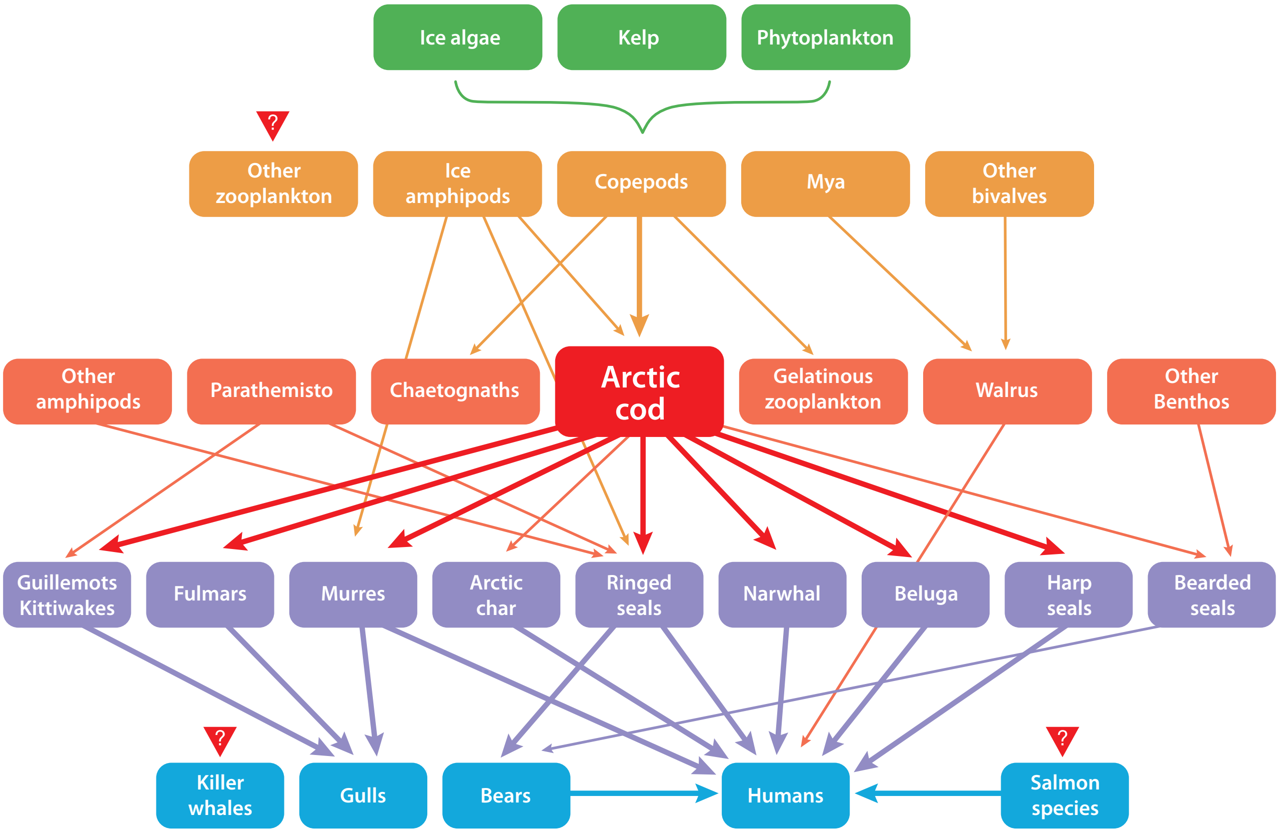

Figure A6.5 Arctic marine food web centered around the key fish species Arctic cod (*Boreogadus saida*). Diagram adapted from Welch et al. (1993). Arrows indicate main energy transfers. For some species insufficient information is available (e.g., other zooplankton) and interaction with potential new species (e.g., salmon species and killer whale) is not yet clear.

Table A6.1 Summary of species temperature and pH threshold information for selected Arctic zooplankton and forage fish species and Arctic char (Steiner et al., 2018). Notes: Adult life stage is assumed when not specified. Limits include sub-chronic thresholds termed 'pejus' limits - tipping points at which performance begins to decline. The Aquatic Species Physiological Limits Database (ASPL Database) will be made public as an evolving database via Fisheries and Oceans Canada.

Species common name:	Temperature limits (°C)			
	Critical lower	Lower pejus	Upper pejus	Critical upper
Pacific herring		Adults = 4.7C		Spawning = 10C Larvae= 13.3C (at 25ppm)
Spot prawn	Adults = 3C	Adults = 5C	Eggs = 13-15C	Adults = 21C Larvae = 15.2C
Arctic cod	-1.4C	Adults = 0.2C	Spawning = 3.5C Adult heart rate (T_{max}) = 10.8C	Adult Loss of equilibrium (LOE) = 14.9C
Arctic char	0C	Eggs < 3C	Adults = 16C Adults Growth freshwater = 15.1C T_{max} (heart rate) = 23C	larvae feeding = 22C Alevins, fry and parr (acclimation 5C) = 23·3, 25·1 and 25·7C
Capelin	-1.5C	0C	Juveniles= 10C Adults= 14C	
Calanus glacialis (copepod)			Stage IV = 10C or = 5C with 3000 µatm added stress	
Limacina helicina (pteropod)	Adults = -1.9C	Adults = -0.4	Adults = 7C	

Franklin et al., 2013). Changes in absolute aerobic scope under stress can affect whole animal health performance. Stressor-related change in heart rate is another reliable index of animal health (Casselman et al., 2012). Hence, laboratory multiple-stress tests based on quantifying the change in cardio-respiratory performance can detect sub-chronic levels of stress, which can better equate to real life applications rather than just testing extreme end points (i.e., mortality). Additional quantification of changes to fatty acid and stable isotope profiles based on acclimation conditions can be used to assess changes in food web dynamics (Budge et al., 2008; Graham et al., 2014; Kohlbach et al., 2017).

Recent observations that Arctic cod distribution is starting to shift northward adds evidence to the model predictions for the future of Arctic cod in the rapidly warming Arctic (Cheung et al., 2008). Despite this potential threat to an entire marine ecosystem and the peoples that depend on it, there are few laboratory-based thermal tolerance studies on Arctic cod and even fewer quantify the impact of multi-stressors (Kunz et al., 2016; Leo et al., 2017). Interestingly, Drost et al. (2016) showed that after one month acclimation at 6.5°C, Arctic cod can actually improve their cardio-respiratory performance. Clearly this species must have some capacity for thermal acclimation (the ability of a species to adjust its optimum performance to a change in temperature) and the greater than expected thermal performance of larvae and adult Arctic cod suggests that temperature may not be the sole driver of their biogeographic range. It appears that the under-ice ecosystem with which they are associated serves as a critical component of their biographic distribution and survival.

A6.3 Methodology

Figure A6.1 shows the various tools of the multi-step framework and their interlinkages. This section provides more specific details about the tools applied in this study.

Climate model projections

Global Earth System Models

Projecting future ecosystem responses to climate change, ocean acidification and other potential stressors requires the application of numerical ecosystem models. The 5th Coupled Model Intercomparison Project (CMIP5) includes a variety of Earth System Models (ESMs); model systems with fully-coupled atmosphere, ocean, sea ice, and land components including interactive biogeochemical modules for all components. ESMs make it possible to study how biogeochemical cycling could respond to transitions in temperature, vertical stratification, sea-ice retreat, and ocean acidification as a first step toward addressing consequences for socio-economic activities (e.g., Deal et al., 2013; AMAP, 2017).

A multi-model comparison of ocean acidification projections using the Intergovernmental Panel on Climate Change (IPCC) representative concentration pathway (RCP) scenarios, specifically RCP8.5 (business-as-usual, reaching >1370 ppm atmospheric CO_2-equivalent in 2100, Moss et al., 2010) is presented by Steiner et al. (2014) and selected results are included here. The intercomparison includes six ESMs: Canadian ESM version 2.0 (CanESM2), Geophysical Fluid

pH	Vulnerable life stage to increasing water temperature and ocean acidification	Key sources
Critical/Pejus/Optimum		
	TBD	Alderdice and Velsen, 1971
	TBD	King, 1997
No difference in proton leak and ATP production efficiency between groups acclimated at 400 µatm vs. 1170 µatm of CO_2	1+yr during surface swarming spring and summer and during diel water column migrations for food. NB: Spawning and egg development occur in winter under ice	Drost et al., 2014; Kent et al., 2016; Laurel et al., 2015; Kunz et al., 2016; Leo et al., 2017
Pejus = CO_2 <10 mg/L optimum 10-20 mg/L	TBD	Sutterlin and Stevens, 1992; Thyrel et al., 1999; Quinn et al., 2011; Musa, 2013; Penney et al., 2014; Hansen et al., 2017; Jeuthe et al., 2016
	TBD	Fortier and Leggett, 1985; Rose and Leggitt, 1989; Rose, 2005
Hatching delay: 6.9 pH Copepodites stage IV: 7.87 pH	TBD	Weydmann et al., 2012; Hildebrandt et al., 2014; Thor et al., 2017
Adult: 28% decrease in calcification at 760 µatm. $\Omega \sim 0.8$ = severe damage	TBD	Comeau et al., 2009; Bednaršek et al., 2012

Box A6.1 Domain and resolution for NAA-CMOC

To drive future projections under RCP8.5 for the period 2006–2085, the 22-km resolution Canadian Regional Climate Model version 4 (CanRCM4) (Scinocca et al., 2016) covering the Coordinated Regional Climate Downscaling Experiment (CORDEX) Arctic domain was merged with CanESM2 where CanRCM4 does not fully cover the North Atlantic Arctic domain to create a forcing data set for the NAA-CMOC model. Daily surface wind forcing was applied as is, while anomalies were calculated for all other variables: daily snowfall, air temperature (at ~10m), longwave radiation, shortwave radiation, liquid precipitation and specific humidity. The anomalies were calculated by combining the merged CanESM2-CanRCM4 dataset with the Coordinated Ocean-ice Reference Experiments (CORE2) dataset[1] (Large and Yeager, 2004, 2009) for the reference period 1986–2005, following the approach of Dumas et al. (2006). The initial ocean temperatures and salinities are set from PHC3.0[2] (updated from Steele et al. 2001) and the initial values for dissolved inorganic carbon (DIC), alkalinity and nitrate were taken from annual GLODAP v2b climatology data (1972–2013)[3] (Lauvset et al., 2016). Other biogeochemical open boundary fields were assigned constants. Ice thickness, ice concentration and snow thickness were initialized from a five-year run with repeated 2006 anomaly forcing. River runoff was derived from observations (Dai and Trenberth, 2002; Dai et al., 2009) with river DIC from Tank et al. (2012) representing coastal and river runoff (no changes were applied for future time periods). Monthly time series open boundary conditions were derived from CanESM2 model output by linearly interpolating between the climatological means for the periods 1986–2005 (representing 1996), 2006–2025 (representing 2016) and 2066–2085 (representing 2076).

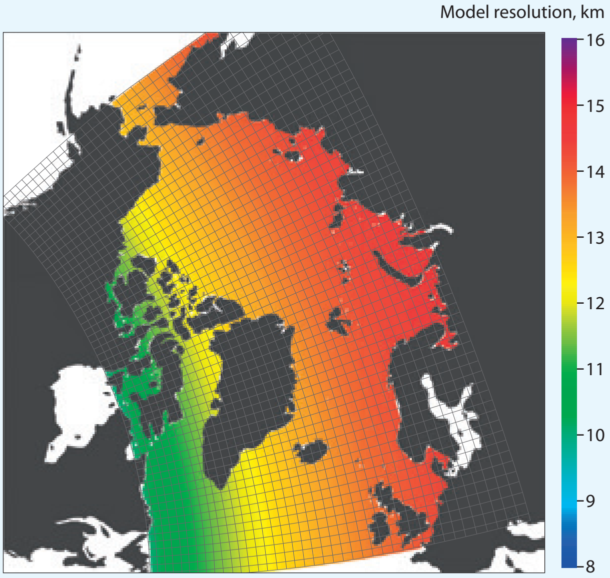

Figure A6.6 Model resolution of the North Atlantic Arctic Model (Hu and Myers, 2013). Color barn indicates model resolution in km, showing highest resolution in the Canadian Arctic.

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[1] http://data1.gfdl.noaa.gov/nomads/forms/core/COREv2/code_v2.html

[2] http://psc.apl.washington.edu/nonwp_projects/PHC/Climatology.html

[3] http://cdiac.ornl.gov/oceans/GLODAPv2/

Dynamics Laboratory ESM (GFDL-ESM2M), Met Office Hadley Center ESM (HadGEM2-ES), Institut Pierre Simon Laplace low-resolution ESM (IPSL-CM5A-LR), Japan Agency for Marine-Earth Science and Technology ESM (MIROC-ESM), and Max Planck Institute for Meteorology low-resolution ESM (MPI-ESM-LR). A subset of these models (higher resolution versions GFDL-ESM2G, IPSL-CM5A-MR, MPI-ESM-MR) was used for the species modelling component of this assessment.

The regional Arctic ocean ecosystem model

The still fairly coarse horizontal and vertical resolution of ESMs restricts the ability to resolve biological or chemical processes happening in the euphotic zone as well as small-scale physical processes important for biogeochemistry. Hence regional climate models (RCMs) and basin-scale models need to be included. The latter are available to a limited extent so far and only one example is used here. While these models improve applicability to local scales, results from individual models come with a greater range of uncertainty.

The North Atlantic Arctic - Canadian Model for Ocean Carbon (NAA-CMOC) is a regional model for the Arctic with highest resolution in the Canadian Arctic (see Box A6.1). The physical model is based on NEMO-LIM2 in the configuration of Hu and Myers (2013, 2014). The ecosystem component CMOC (Zahariev et al., 2008; Christian et al., 2010) is the same ecosystem model applied in CanESM2. The application in the Arctic shows good correspondence with observed chlorophyll features in the Canada Basin (Steiner et al., 2016). The model was forced with RCM output to simulate the period 2006–2085. Details on the forcing and input data are provided in Box A6.1.

The species distribution model

Species distribution models can be used to project how species distribution and abundance could change under different climate trajectories. A dynamic bioclimatic envelope model (DBEM) developed by Cheung et al. (2008, 2011, 2016) was applied to project how marine species could respond to changes in environmental conditions in both time and space using mechanistic and correlative processes. The DBEM uses an integrative approach, linking ESMs, species distribution models (Close et al., 2006), advection-diffusion movement models (Sibert et al., 1999), growth models (Pauly, 1980), physiological models (Pauly, 1981; Pörtner, 2010) and population dynamics models (Pauly, 1980; Hilborn and Walters, 1992; O'Connor et al., 2007).

Initial species distributions (Close et al., 2006) were used to characterize species-specific habitat suitability by overlaying environmental variables such as temperature, salinity, depth, sea ice, and dissolved oxygen concentration. Species-specific habitat preference parameters were included to create a bio-envelope and habitat preference profile for each species

(Cheung et al., 2008). Individuals grow using a von Bertalanffy model and are constrained by ecophysiological conditions such as temperature, oxygen availability and salinity (von Bertalanffy, 1951; Cheung et al., 2011). Population growth is represented by a logistic growth function (Hilborn and Walters, 1992). Geographical movement of species through time and space is modeled using advection-diffusion models and the relative dispersal rate of each species (e.g., large-bodied pelagic species, small reef-dwelling species) to neighboring cells based on habitat suitability.

Using the DBEM, individual and population growth changes are determined based on environmental parameters, while spatial movement is determined by habitat preference and suitability. Thus, populations move based on gradients of environmental suitability and carrying capacity, and populations can be expected to shift in space and time to optimize area where habitat is more suitable and potential for population growth is higher (for details see Cheung et al., 2008, 2011, 2016). Changes in distribution and abundance for each species were used to estimate changes in fisheries catch potential, which assumes that annual catch potential is equal to the maximum sustainable yield. The DBEM results were used to assess downstream effects on Arctic communities as drivers of socio-economic indicators such as food security (i.e., catch potential) and revenue (i.e., landed value).

Physiological models of responses to environmental change

The DBEM integrates mechanistic models to quantify physiological responses to environmental change on individuals and at the population level. The model assumes that physiological stressors, such as deviations from optimal temperature, act to reduce aerobic scope and thus energy available for growth (Pörtner et al., 2010). The model is grounded on the theory that oxygen supply is limiting post-larval fish growth and reproduction for marine fish and invertebrates and that changes in environmental conditions can alter oxygen supply-demand dynamics and various life history processes and traits (Cheung et al., 2011; Pauly and Cheung, 2018). The DBEM simulates physiological responses to three main environmental drivers: temperature, dissolved oxygen concentration, and acidity (for details on methods see Cheung et al., 2011).

Growth rate over time is determined by the difference between anabolic and catabolic processes. These processes are dependent on the availability of oxygen and oxygen uptake ability. Oxygen supply becomes increasingly limiting as marine organisms grow, due to the lower rate of increase of respiratory surfaces relative to the increase in body size (Pauly, 1981; Cheung et al., 2011; Pauly and Cheung, 2018). The difference between oxygen supply and demand determines aerobic scope. Aerobic scope is further limited by deviations in temperature, and decreases as temperatures reach an organism's upper and lower thermal limits. Ocean acidification and changes in dissolved oxygen concentration were modeled to act synergistically with temperature and shrink the overall aerobic scope (Pörtner and Farrell, 2008; Cheung at al., 2011). Ocean acidification was also modeled as a correlative process to affect survival (Tai et al., 2018). Effect sizes in response to ocean acidification were taken from a recent meta-analysis (Kroeker et al., 2013). The DBEM assumes that species within a taxonomic group (i.e., crustaceans, mollusks) share the same sensitivity to ocean acidification, and so the same mean values were used for each taxonomic group. Effect sizes represent the percentage change in growth or survival for every 100% increase in hydrogen ion concentration as a linear relationship from initial conditions.

Arctic regions and species included in study

Marine regions across the Western Arctic were modeled for this study, including the Chukchi Sea, Beaufort Sea, Canadian Polar Shelf, and Baffin Bay, which together represent over 4 million km^2 of ocean area. A total of 82 marine species were modeled: 57 finfish species and 25 invertebrate species. These are exploited species in some ocean regions with fisheries catches reported in the Sea Around Us marine fisheries catch database (Pauly and Zeller, 2015). Species-specific data to parameterize the DBEM were collected from FishBase (Froese and Pauly, 2017), SeaLifeBase (Palomares and Pauly, 2018) and the ASPL database (Steiner et al., 2018). Invertebrates (i.e., crustaceans and mollusks) were modeled to respond to changes in acidity, in addition to other climate-related drivers such as temperature and oxygen concentration. Meta-analyses have shown negligible effects of acidification on finfish species, but most invertebrate species groups showed some level of sensitivity (Kroeker et al., 2013).

As previously mentioned, Arctic cod are an ecologically and energetically important link in the Arctic food web and particular focus was given to modelling Arctic cod. This study used an ocean acidification effect size of 20% on growth and survival for every 100% increase in acidity. There is currently little research on the effects of ocean acidification on Arctic cod, but there is evidence of negative effects on growth and survival in other marine fish (Munday et al., 2009; Baumann et al., 2012). These effect sizes were chosen to test the sensitivity of Arctic cod abundance to low and high effects of ocean acidification, although the exact sensitivity to growth is not currently known. Parameter uncertainty was tested using low (10%), medium (20%), and high (30%) effect sizes in response to ocean acidification for both growth and survival. Sensitivity to structural uncertainty comparing ocean acidification effects on growth, survival, and growth and survival together were also tested. Finally, comparisons were made between effects on survival for adults and larval stages.

Environmental input data

Model simulations were run using environmental input data from three ESMs (GFDL-ESM2G, IPSL-CM5A-MR, MPI-ESM-MR). These models differ in terms of their parameterizations and structure resulting in dissimilarities in key biophysical drivers, such as primary production (Stock et al., 2011; Vancoppenolle et al., 2013) and acidification (Steiner et al., 2014). With each ESM, two contrasting RCPs were used to evaluate the biophysical responses of marine organisms. The first (RCP8.5) is the 'business-as-usual' trajectory and the associated radiative forcing is projected to increase global sea surface temperature by 3.5°C by the end of the century relative to the pre-industrial period, while the second (RCP2.6) conforms to the Paris Agreement targets of limiting temperature rise to 1.5°C. In this study, the RCP8.5

and RCP2.6 scenarios are referred to as the high and low carbon dioxide (CO_2) scenarios, respectively.

The Beaufort Sea food web model

EwE is a freely available ecosystem modelling software suite designed to improve understanding of complex marine ecosystems. It has three main components: Ecopath (a static, mass-balanced snapshot of the system), Ecosim (a time dynamic simulation module for policy exploration), and Ecospace (a spatial and temporal dynamic module primarily designed for exploring impact and placement of protected areas) (Christensen and Walters, 2004; Christensen et al., 2007). EwE makes it possible to address ecological questions such as ecosystem effects of fishing, management policy options, impact and placement of marine protected areas, and cumulative ecosystem effects of environmental change. EwE is set to a specific study area and fitted to scientific trend data or observations to ensure known changes are captured in the area. The model applied here encompasses the Beaufort Sea marine ecosystem area ranging from 67.5° to 75° N and 112.5° to 158° W (which encompasses the Inuvialuit Settlement Region and the eastern part of the Alaskan Beaufort Sea), which includes estuarine, coastal, and oceanic habitats ranging from 0 to 3000 m of water depth and is described in detail by Suprenand et al. (2018). The applied Ecopath model considers 36 functional groups, which include single species and aggregated groups of species. These functional groups range from top predators (marine mammals) to primary producers and detritus, covering taxa throughout the food web. A list of species for aggregated functional groups, from birds to fish, are provided by Suprenand et al. (2018: their Table S1). An Ecopath model broadly represents an instantaneous 'snap-shot' of material fluxes in the ecosystem according to the constraints of mass-balance and the conservation of energy. For each functional group, production is determined for all components of the food web and linked through diet proportions. In general, energy must be produced by each group to balance energy removed through predation, fishing, migration, and other mortality under the mass-balance assumption. In estimating the dimensionless trophic levels, EwE assigns primary producers a level of one, whereas all other functional groups have a trophic level of one plus the diet-weighted average of their prey item's trophic levels. For trophic level comparisons across all studies and to functional groups, single species were aggregated into similar functional groups with a mean trophic level.

As Ecopath predator-prey relationships (trophodynamics) define the movement of energy throughout the ecosystem, a methodology described by Clarke et al. (2008) was applied to perform similarity and dissimilarity profile analyses (Suprenand et al., 2018). These result in hierarchical clusters of predator groups according to their similarities in prey, and prey with similarities in their predators. In general, hierarchical clusters reveal similarities among and between functional groups (or species) with regard to energy consumption by predators or energy contributions from prey, respectively. The analyses are used to reveal which functional groups are most significant in structuring energetic pathways of the Beaufort Sea marine ecosystem. While Arctic cod provide a central pathway in the food web, ecosystem analyses reveal arthropods, ringed seal (*Pusa hispida*) and polar bear (*Ursus maritimus*) as potential keystone species in the food web.

While Ecopath provides an instantaneous snap-shot of the energy balance between predator-prey relationships, the Ecosim model performs temporal simulations, with impacts on species groups including direct impacts (harvest and predation mortality) and indirect impacts (environmental changes or shifts in prey items). Ecosim adds a temporal dimension for predicting biomass changes in primary producers and consumers. Adding layers to account for harvest and environmental changes allows users to examine responses by functional or species groups. The foraging arena theory dictates that the biomass of a group is split between vulnerable and invulnerable states, whereas the prey is only vulnerable to predators during the vulnerable state (Walters et al., 1997). Biomass trends, harvest trends, fishing effort, and forcing and mediation functions (environmental drivers) have been developed for the Ecosim model, with vulnerability for each functional group developed for the Ecosim model. The vulnerability for each functional group is determined when fitting the model to a time series. As the model is validated, it can be used to run future simulations taking into account a range of potential environmental changes. These simulations will be used to assess the economic impacts on communities in the region under different future emission scenarios.

The economic model

Fisheries harvest in the Canadian Arctic is largely for subsistence purposes, providing key sources of food and nutrition for Inuit communities (Zeller et al., 2011). Arctic commercial fisheries are limited in Canada, but the increase in accessibility with ocean warming and decreasing sea-ice cover may encourage their development. Outputs of catch potential from the DBEM were used to assess the potential for Arctic fisheries at present (2001–2010) and in the future (2091–2100). Ex-vessel prices (i.e., the price of catch at the first point of sale) for each of the species explored in this analysis were used to estimate the potential value of present and future Arctic fisheries. Prices were weighted and averaged across the countries that reported catch for each species and averaged for the 2001–2010 period. For ease, current prices were carried forward to estimate future fisheries landed values. Prices were taken from a global ex-vessel price database (Sumaila et al., 2007; Swartz et al., 2013; Tai et al., 2017) (available at www.seaaroundus.org).

A6.4 Results

Recent environmental change in the Western Canadian Arctic

In the Arctic, ocean acidification is intensified due to low temperatures, increased freshwater supply (river runoff and ice melt), riverine carbon influx and low pH Pacific water inflow, all of which are affecting the Western Arctic Bioregion. (Note that pH is commonly used as a measure for the status of acidification, while the saturation states of aragonite and calcite – Ω_{arg} and Ω_{cal} – are frequently used to indicate the

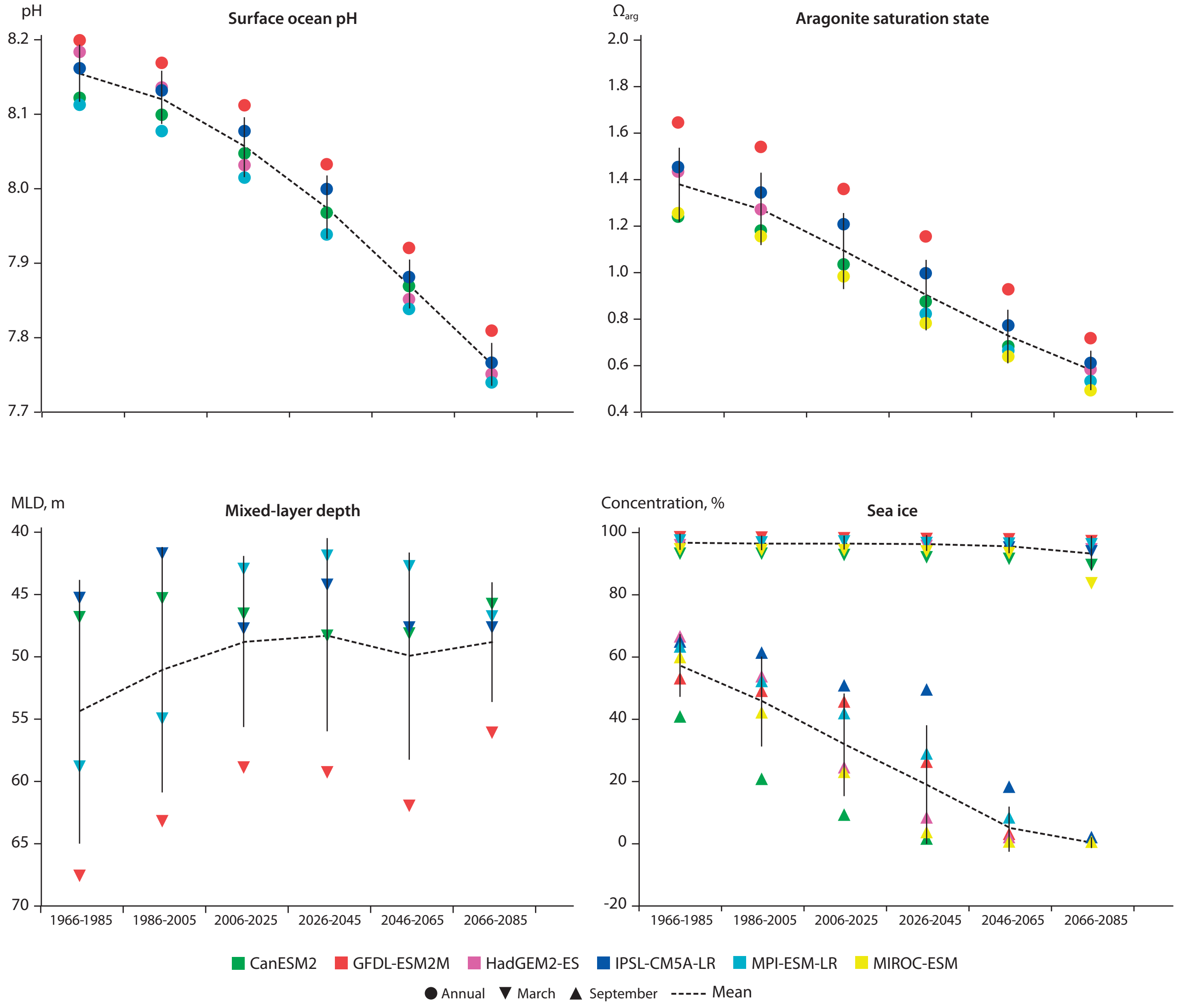


Figure A6.7 Multi-model comparison of bi-decadal averages for Canada Basin (68°–79°N, 124°–160°W). Plots show projected change in annual mean surface ocean pH, annual mean aragonite saturation state, maximum mixed layer depth in March, and sea ice concentration in March and September for RCP8.5. Shown are individual model averages as well as the multi-model mean and standard deviation. Note that pH is not available for MIROC-ESM and maximum mixed layer depth in March is not available for MIROC-ESM and HadGEM2-ES. Based on Steiner et al. (2014).

threat to marine species). Large gradients in surface pH and Ω_{arg} over small spatial scales, particularly strong in sea-ice covered areas arise due to spatial variability in primary production and remineralization processes (Tynan et al., 2016) with additional reduction in under-ice pH due to the release of CO_2-enriched brines when ice forms. Chapter 2 provides a more detailed description of recent changes in the area. In summary, observations in the Western Canadian Arctic indicate a significant northward expansion of low Ω_{arg} waters with upper halocline waters and deep waters now regularly showing aragonite undersaturation particularly in the off-shelf waters (Miller et al., 2014; Qi et al., 2017, see Chapter 2, Figure 2.4). These results further confirm that more rapid acidification is occurring in the Arctic Ocean than the Pacific Ocean and Atlantic Ocean, with the western Arctic Ocean representing the first open-ocean region with large-scale expansion of 'acidified' water directly observed in the upper water column. These changes in seawater chemistry are propagated via M'Clure Strait and Amundsen Gulf into waters of the Canadian Polar Shelf and beyond (AMAP, 2013).

Projected change in the Western Canadian Arctic

Projections performed with ESMs consistently show enhanced ocean acidification in polar regions and suggest Arctic Ocean acidification will continue over the next century with accelerated reductions in calcium carbonate ($CaCO_3$) saturation state at least until the sea-ice cover reaches a new steady state with largely ice-free summers (Steiner et al., 2014, see Chapter 2, Figure 2.7). Projections with RCP8.5 forcing for Canada Basin show reductions in bi-decadal mean surface pH from about 8.1 in the period 1986–2005 to 7.7 by 2066–2085, closely linked to reductions in $CaCO_3$ saturation state from about 1.4 to 0.7 for aragonite and 2.0 to 1.0 for calcite. The changes coincide with a decrease in sea-ice concentration and maximum mixed layer depth (*Zm*) (Figure A6.7).

Projections performed with higher resolution ocean-only models provide more detailed projections for the region, while showing very similar trends and pan-Arctic patterns for

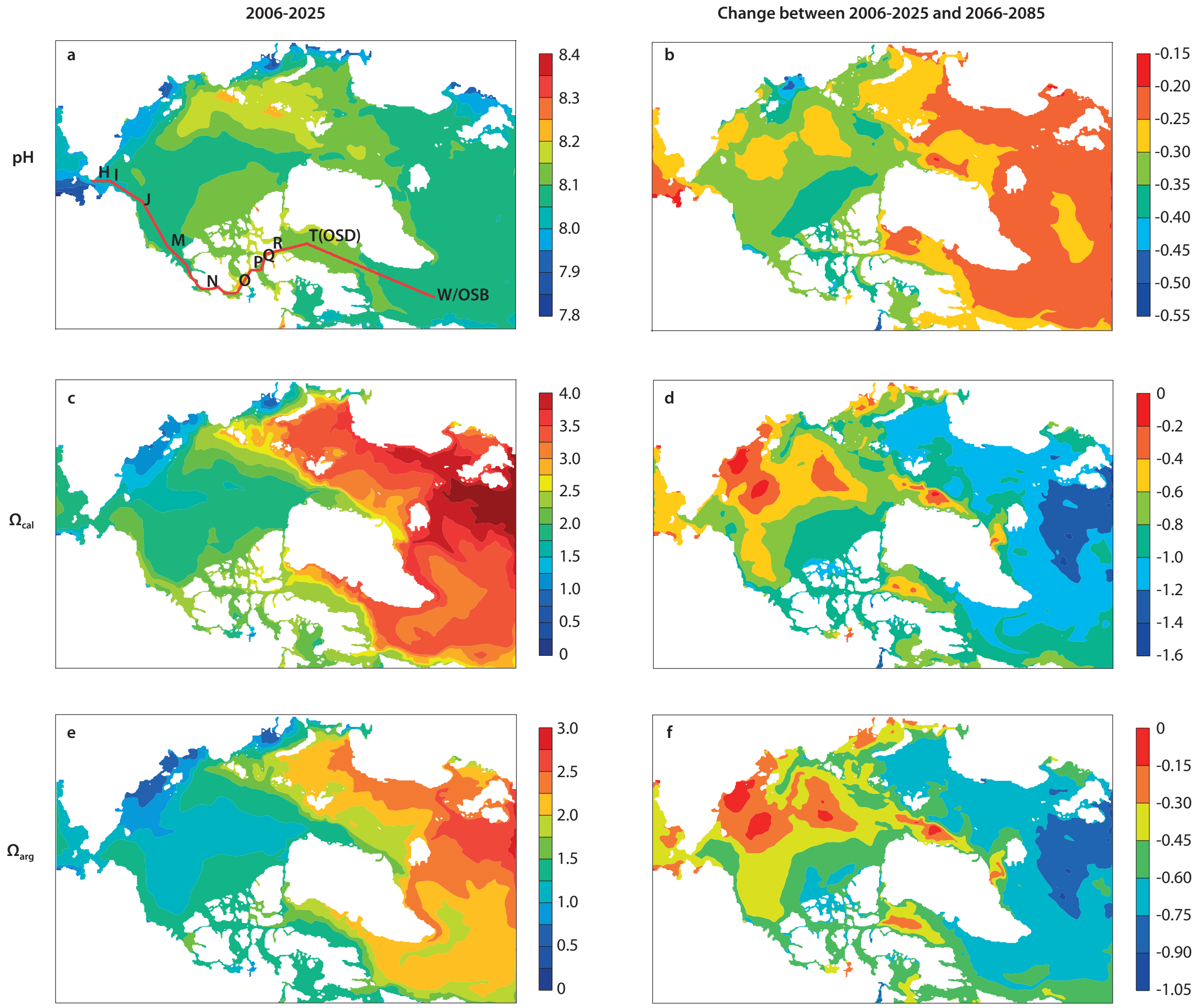


Figure A6.8 Simulated summer (July, August, September) surface pH, calcite saturation state Ω_{cal} and aragonite saturation state Ω_{arg} for the bi-decade 2006–2025 and projected change for the period 2006–2025 to 2066–2085 under the RCP8.5 scenario.

pH (Figure A6.8a versus AMAP, 2013) and Ω_{arg} (Figure A6.8e versus CanESM results in Chapter 2, Figure 2.7). Figure A6.8 shows the projected change (2066–2085 minus 2006–2025) for pH, Ω_{cal}, and Ω_{arg} under the high CO_2 emission scenario (RCP8.5), with decreases in the Western Arctic Bioregion of -(0.3-0.4) for pH, -(0.6-0.9) for Ω_{cal}, and -(0.4-0.6) for Ω_{arg}. This indicates aragonite undersaturation for the whole region while surface waters remain supersaturated with respect to calcite. Figure A6.9 shows cross-sections from Bering Strait through the Canadian Polar Shelf to Baffin Bay for pH, Ω_{arg} and Ω_{cal}, and temperature (summer averages, JAS) as simulated with the NAA-CMOC model under RCP8.5. AMAP (2013) indicated that changes in $CaCO_3$ saturation state in Arctic surface waters propagate downstream from the Beaufort Sea through the Canadian Polar Shelf southward through the Labrador Sea and that a subsurface core of low-saturation state waters can be traced across the Canadian Polar Shelf and into Baffin Bay to Davis Strait (AMAP, 2013: Figure 2.13). The projections show a continuous decline across all depths over time for both Ω_{arg} and Ω_{cal} and the low saturation core merging with undersaturated bottom waters within the central Canadian Polar Shelf. By 2066–2085 all of the central Canadian Polar Shelf is projected to be undersaturated with respect to Ω_{arg}, and bottom water undersaturation is reached for Ω_{cal}. This is also apparent in Figure A6.10 which shows the regionally averaged progression of Ω_{arg} and Ω_{cal}. The enhanced acidification coincides with a continuous rise in temperature across all depths (not shown). Acidification advances faster on the western side of the Canadian Polar Shelf, including the Western Arctic Bioregion and is projected to reach calcite undersaturation for near surface waters by the end of the century.

The model simulations suggest a combination of stressors potentially affect the performance, abundance and phenology of marine species. Ice thickness is simulated to decrease at a rate of -(0.15–0.2)m/decade for all seasons, with the strongest declines in winter (Figure A6.11). Ice concentration is projected to decrease most in summer (-7%/decade), and to a lesser extent in winter (-2%/decade; not shown), which might have both positive and negative effects on species due to effects on habitat and primary production.

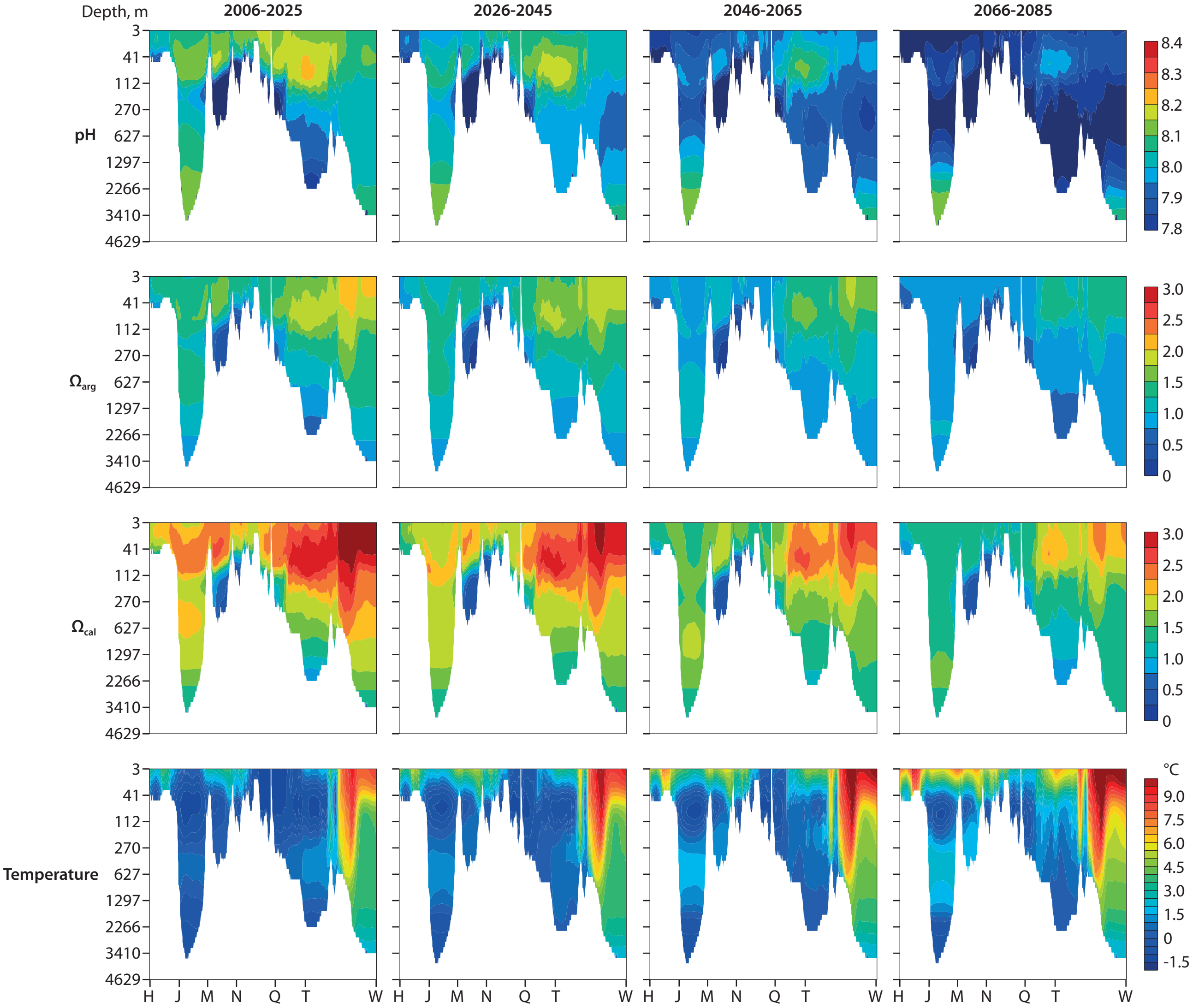
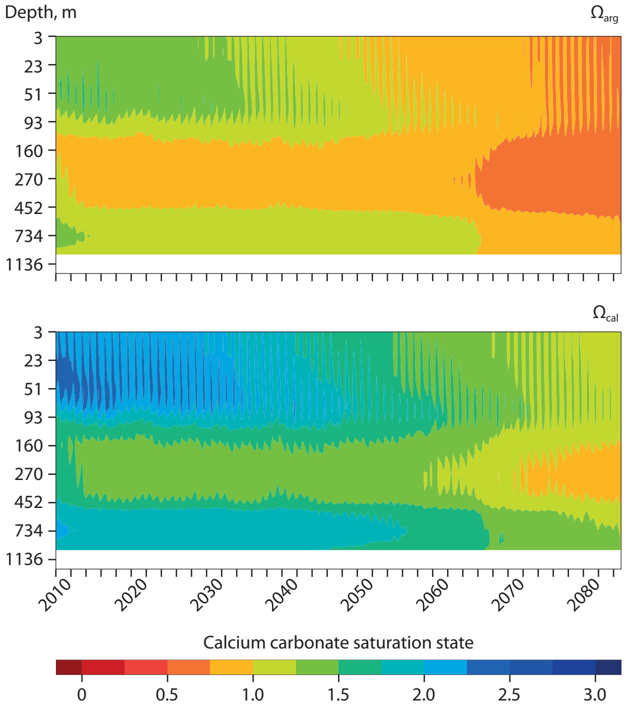
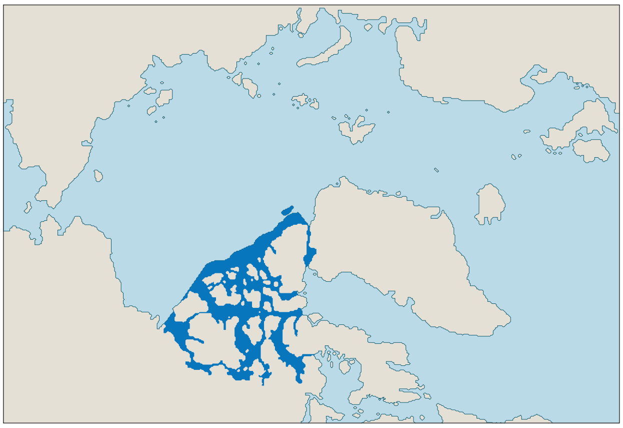


Figure A6.9 Projected change under the RCP8.5 scenario for pH, calcium carbonate saturation state and temperature in summer (July, August, September) along the transect through the Canadian Polar Shelf shown in Figure A6.8a. y-Axis indicates mid-level depth of each model level.





🟦 Canadian Polar Shelf, area over which the data have been averaged

Figure A6.10 Temporal evolution of calcium carbonate saturation state (Ω_{arg} and Ω_{cal}) as simulated with the NAA-CMOC model. Data shown represent a regional average for the Canadian Polar Shelf, indicated on the map. y-Axis indicates mid-level depth of each model level.

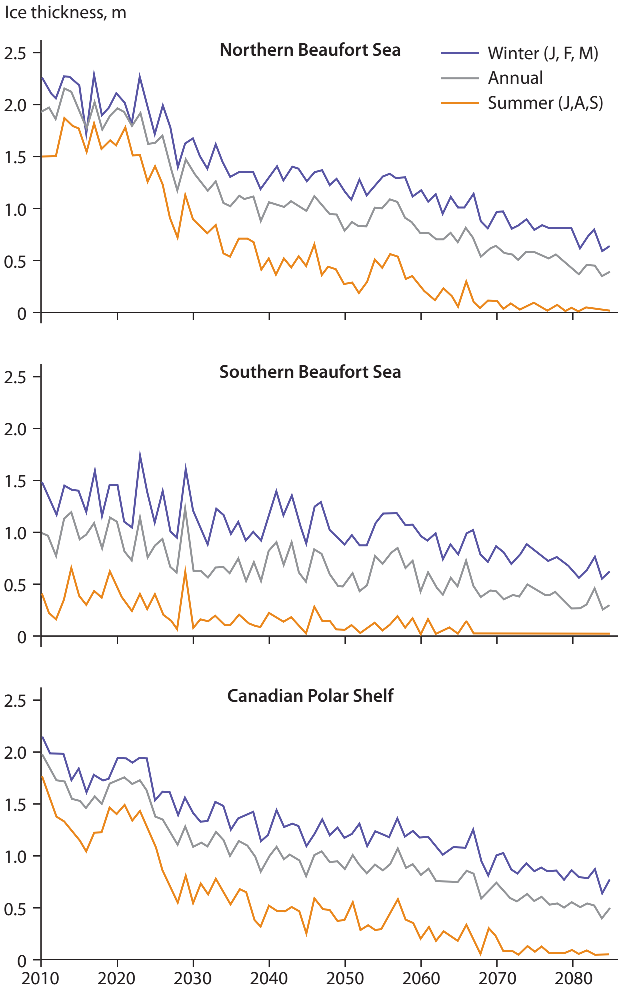


Figure A6.11 Simulated temporal evolution of seasonal ice thickness averaged over the northern Beaufort Sea, southern Beaufort Sea, and Canadian Polar Shelf.

Ecopath model results – recent changes in the Beaufort Sea

At present, the Ecopath model has been validated to past trends, but has not yet been directly linked to the climate model projections or to the DBEM and acidification impacts. Nevertheless, this initial analysis already indicates the occurrence of food-web shifts and tipping points relevant to Arctic cod and higher trophic level species.

The EwE analysis (Suprenandet al., 2018) indicates that over the past 45 years, predators are progressively diversifying their diets, depending less on traditional predator-prey relationships, and increasing their intake of prey groups once less frequently consumed. Consequently, prey groups have been reducing their energetic contributions to specialized predators, but providing more energy to a greater number of predators overall. However, this trend towards predatory generalists probably minimizes more specialized and/or highly co-evolved trophodynamic exchanges within the Beaufort Sea marine ecosystem. These food-web structural changes led to clear reductions in similarities in prey hierarchical clusters between 1970 and 2014, suggesting that Beaufort Sea trophodynamics have been distinctly restructured over the past 45 years. The main drivers of change for Beaufort Sea marine trophodynamics are bottom-up forcing and mediation functions expressing decreases in sea-ice extent and increases in sea-surface temperature. These are in addition to subsistence harvests of relevant species.

Examples of food-web structural changes that affect keystone functional groups include increases in the biomass of arthropods, micro-zooplankton, and other meso-zooplankton, which are primarily impacted by increased primary production and pelagic detritus. A small reduction in the bivalve group's biomass follows a decrease in benthic detritus. Benthic detritus originates primarily from sea-ice algae and the organic matter forms larger aggregates and sinks faster under sea ice (Michel et al., 2002; Juul-Pedersen et al., 2008). Changes in key groups lead to an increase in competition between walrus (*Odobenus rosmarus*) and gray whales (*Eschrichtius robustus*) for bivalves, an increase in competition between ringed seals and several other Beaufort Sea predators for Arctic cod, and polar bears eating less of their traditional prey, ringed seals, and increasing their incidents of cannibalism and potential foraging outside of the model area.

Suprenand et al. (2018)'s operating model trophodynamics indicate that the annual growth of phytoplankton biomass (+23% increase from 1970 to 2014) will benefit zooplankton (+18%; all groups), yield mixed impacts on invertebrates (+11%), and benefit three species of marine mammal: beluga (*Delphinapterus leucas*) (+56%), gray whale (+18%), and spotted seal (*Phoca largha*) (+16%). While increases in primary production also benefit the keystone functional groups of arthropods, micro-zooplankton, other meso-zooplankton, and Arctic cod (e.g., Michaud et al., 1996), either directly as prey and/or indirectly through the diets of their prey, the benefits to Arctic cod are likely to be limited. With increasing sea surface temperature, Arctic cod may reach its upper thermal limit (Graham and Hop, 1995; Drost et al., 2014) thereby reducing suitable Arctic cod habitats in the Beaufort Sea, and inhibit their energetic contribution to the entire Arctic food web (e.g., Nahrgang et al., 2014).

EwE results suggest Arctic cods' biomass is positively influenced by small primary producers (<5 μm), micro-zooplankton, and the other meso-zooplankton functional group. While Arctic cod consume these functional groups, polar bears act to limit ringed seals' predatory pressure on Arctic cod, thus yielding an indirect positive influence overall. According to the operating model's historic reconstruction from 1970 to 2014, ringed seals (whose diet is 12% Arctic cod) are becoming increasingly reliant on their Arctic cod prey, which constitute a larger percentage of their overall diets. The ringed seal functional group, also has a direct influence on the health of the polar bear functional group. If a sudden decrease in the energetic contributions from Arctic cod occurs as the Beaufort Sea warms, higher trophic level predators (beluga, ringed seal, Arctic char) reliant on Arctic cod (Bradstreet et al., 1986; DFO, 1999; Loseto et al., 2009) will have to alter their prey, possibly opening the door for subarctic species such as capelin (*Mallotus villosus*) (Rose, 2005; Walkusz et al., 2013). Capelin are still considered to be of low abundance in the region compared to Arctic cod; however, they co-occur across the Beaufort Sea and have dietary overlaps with Arctic cod (Cobb et al., 2008; McNicholl et al., 2016). Furthermore, while capelin are consumed by marine mammals in other Arctic regions (Bluhm and Gradinger, 2008), their importance to marine mammals (beluga and ringed seal) in the Beaufort Sea was not noted until recently (Choy et al., 2016). Increases in capelin as a forage fish have occurred in Hudson Bay (Gaston et al., 2003) and the Barents Sea (Hop and Gjøsæter, 2013), with the potential to at least partially replace Arctic cod (Hoover et al., 2013a,b), highlighting the potential for large increases in the Beaufort Sea. However, the energetic contributions of Arctic cod and capelin are similar (5–7 vs. 4–5 kJ/g ww respectively according to Hop and Gjøsæter, 2013, with higher values of up to 8.4 kJ/g ww reported by Lawson et al., 1998) meaning energetic impacts from shifting diets to capelin from Arctic cod may be limited, assuming there are no ecological (predator-prey) barriers.

With a diminishing mean monthly sea-ice extent and increasing mean monthly sea-surface temperature, the biomass from 1970 to 2014 increased for the majority of Beaufort Sea functional groups, primarily driven by an overall increase in primary production. The functional group with the single greatest increase is bowhead whale (*Balaena mysticetus*) with a five-fold growth in population, as it is recovering from historical whaling depletion and is considered a rebounding population in the model. This increase, while large in the context of a single species only accounts for a small proportion of the overall ecosystem biomass. However, the fish functional groups, when considered collectively, had the largest biomass increases overall (59%), highlighting the potential for fish groups to increase as the region becomes more productive with a longer ice-free season. One of the issues with the bowhead whales' near-exponential growth over the past 45 years is their potential to reach carrying capacity (Brandon and Wade, 2006). At their carrying capacity, the energetic demand of the bowhead whales functional group as a whole will be the greatest, which will directly impact the biomass of five of the nine key functional groups. Furthermore, as the seasonal sea-ice extent decreases, the Arctic is likely to include more suitable habitats for more whales (bowhead whale, beluga, gray whale, and killer whale *Orcinus orca*), assuming they have enough available prey. This will increase their seasonal residence time, as migrations into the Beaufort Sea are likely to take place earlier in the year, and migrations out of the Beaufort Sea are likely to take place later in the year. The potential for ecological imbalances with the bowhead whales is great when considering resource limitations alone, as more time spent in the region may mean increased reliance on lower trophic level resources. Furthermore, the increase in fish biomass does not necessarily translate into increased biomass of apex predators.

When examining Shannon Index values from 1970 to 2014, the relationships between biodiversity and environmental variables indicate that the Beaufort Sea may already be experiencing food-web structural impacts due to climate change, and tipping points may have already been reached (Lindsay and Zhang, 2005). With the assumption that the 1970 trophodynamics, sea-ice extents, and sea-surface temperatures characterize a period when the Beaufort Sea marine ecosystem was stable, statistical analyses indicate that a significant climatological tipping point was reached in the early 1990s, and a significant biodiversity tipping point was reached in the late 1990s. Following the climatological tipping point, the mean annual whole-ecosystem biomass values and trophic levels significantly increase. And, following the biodiversity tipping point, the Beaufort Sea marine ecosystem displays increasing mean annual evenness (as measured by the Shannon Index). The changes in predator diets, and increase in mid-trophic level biomass, support these observations. However, even as the majority of the Beaufort Sea functional groups have benefited from the increased primary productivity, the model still suggests decreases in biomass for polar bear, walrus, ringed seal, bearded seal (*Erignathus barbatus*), and minor decreases for benthic functional groups since the biodiversity tipping point. This indicates feedback loops through the Beaufort Sea food web that negatively affect almost half of the ecosystem's keystone species. As the other key species mostly benefit from these inter-ecosystem feedback loops and given a keystone species' potential impact to the ecosystem, it is very likely that the entire Beaufort Sea marine ecosystem will continue to undergo significant food-web restructuring in the decades to come, as environmental changes continue.

Suprenand et al. (2018) identified functional groups that, positively or negatively, influence the 2014 biomass of polar bears and bowhead whales. These two species of marine mammal were chosen because they are actively managed by state, local, and/or international governances, their annual biomass estimates are available from 1970 to 2014, and they are influenced by the diminishing sea-ice extent in different ways. Ongoing management of polar bears and bowhead whales in the Beaufort Sea is likely to benefit from increased monitoring efforts of the keystone functional groups; specifically, population assessments and trends for ringed seal, Arctic cod, micro-zooplankton, and small primary producers (<5 μm, potentially captured by spring bloom monitoring efforts in the region).

Historical simulations (1970–2014) reveal changes in species evenness (as measured by the Shannon Index) and have statistically significant associations with sea-ice extent anomalies, sea-surface temperature anomalies, biomass and the mean trophic level before and predominantly after 1992. This temporal transition is a potential 'climatological tipping point' (Lindsay and Zhang, 2005; Suprenand et al., 2018).While these simulations incorporate sea ice and sea-surface temperature, other methods such as mediation and forcing functions are being developed to incorporate the impacts of ocean acidification to address cumulative impacts on the ecosystem.

Annual harvest rates for marine mammals and fish are included as 'fishing' effort per month, defined for each of the Ecopath fisheries, such that each Indigenous community and each functional group harvested by that community are considered in Ecosim temporal simulations (Suprenand et al., 2018).

Based on their model results, Suprenand et al. (2018) suggested that a tipping point in the food-web restructuring has already occurred. However, as Arctic sea-ice extent continues to decline and sea-surface temperature continues to rise, and the population of whales reaches carrying capacity while the biomass of keystone functional groups continues to change, even greater changes to the Beaufort Sea marine ecosystem are likely to occur. An application of the model with projected future climate forcing is under development but was not available for this report.

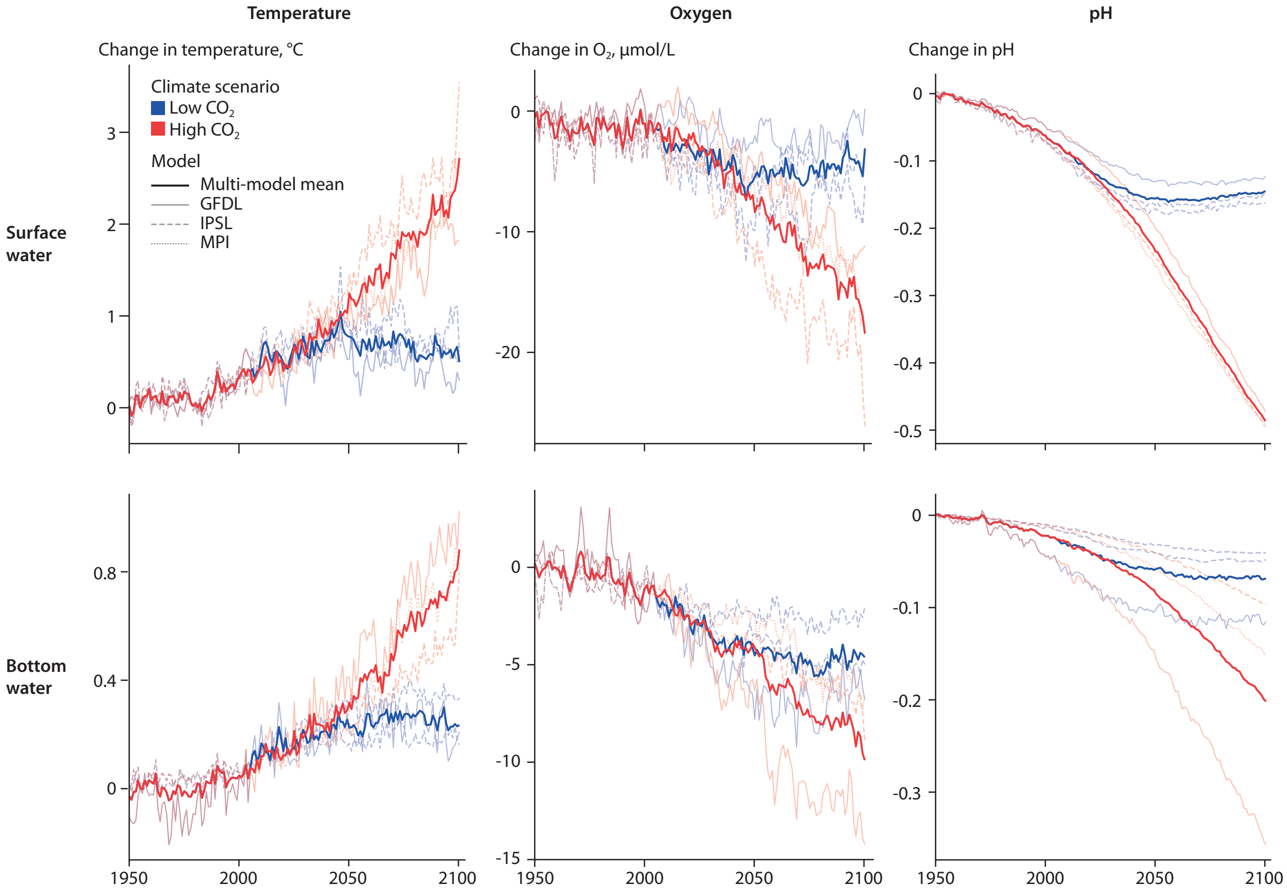


Figure A6.12 Projected changes in sea surface and deep ocean variables using three ESMs (GFDL-ESM2G, IPSL-CM5A-MR, MPI-ESM-MR). Bold lines represent multi-model averages and trajectories are relative to the 1951–1960 average. Trajectories are shown for two climate scenarios: 'Low CO_2' where CO_2 emissions are mitigated and Paris Agreement targets are met (RCP2.6, +1.5°C global sea surface temperature) and 'High CO_2' under 'business-as-usual' (RCP8.5, +3.5°C).

Projected changes in species distribution

Input data for ESMs

Model projections suggest that Arctic marine environments are likely to experience much change. The ESMs used to force the DBEM are consistent in their trajectory of environmental conditions, but show variation in the rate of change (Figure A6.12). Sea-surface temperature is projected to increase by ~2.5°C across the Western Arctic region considered here (note unless otherwise indicated, the DBEM results include the entire area shown in Figure A6.15), with one model projecting an increase of >3.0°C (IPSL). The projections also suggest a decrease in sea-surface oxygen (Figure A6.12), probably due to temperature-related decreases in solubility. Different rates of sea-ice retreat in the region cause differences in stratification and mixing, in turn affecting exchanges with the atmosphere, including heat uptake in summer. All models show reduced vertical mixing in the central Beaufort Sea, but increases to different degrees in the shelf areas (e.g., Steiner et al., 2014). Localized processes are likely to affect coastal regions differently causing some areas to experience greater or lesser changes in environmental conditions than the overall average over the entire Arctic region.

Exploited species

Overall maximum catch potential (MCP) for the 82 modelled species is projected to increase by ~35% by the end of the century (relative to the 2001–2010 period) under the high CO_2 emission scenario (RCP8.5), while the low CO_2 emission scenario (RCP2.6) shows negligible effects (Figure A6.13). While model results from different ESMs show variability the overall trends are distinct and do not overlap. Effects of ocean acidification on invertebrates reduced the projected increase in MCP by >10%, suggesting that ocean acidification minimizes any gains from temperature-driven shifts in distribution. The results show a significant relationship between ocean acidification and decline in MCP for invertebrate species (Figure A6.14). MCP shows little response to small changes in ocean pH with few consequences for the low CO_2 emission scenario, while the high CO_2 emission scenario creates a 'threshold' response where MCP declines sharply after mid-century (2050) as the rate of change in pH increases (see Figure A6.12).

Changes in MCP of all species show increased catch potential across areas of the Canadian Polar Shelf, Baffin Bay, the North Bering Sea, and Chukchi Sea under the high CO_2 emission scenario (Figure A6.15a). Although the results show general increases with higher emissions, the increase for invertebrate species is reduced across the Canadian Arctic due to ocean acidification (Figure A6.15b). Further increases in MCP are due to increased species turnover (i.e., species invasions and extinctions across geographic areas). The results show an increase in species turnover number in the southern Arctic regions, while the rate of turnover is highest in the north

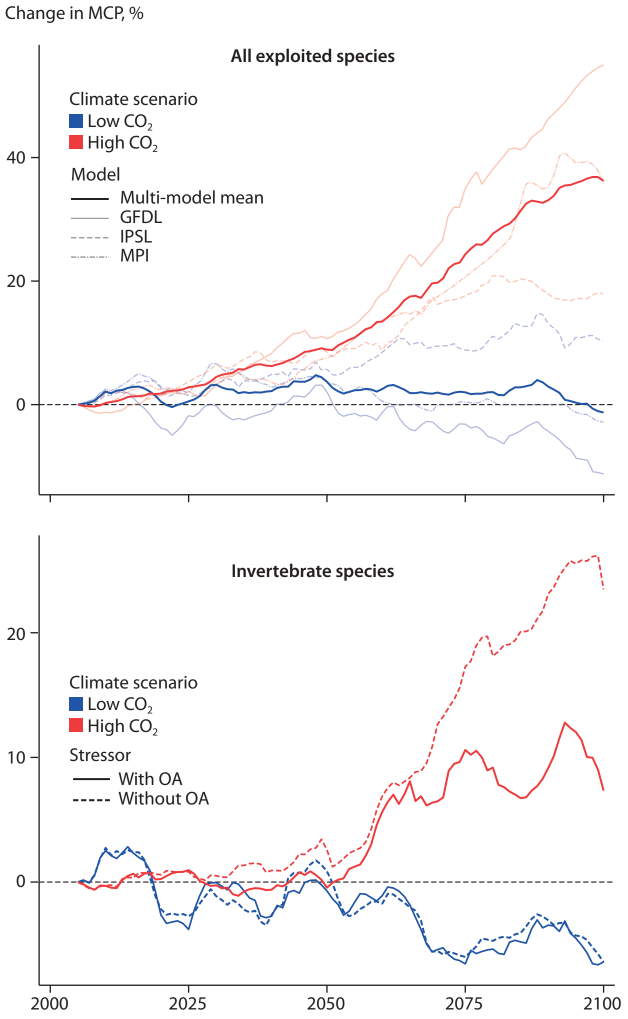


Figure A6.13 Projected changes in maximum catch potential (MCP) for all species in response to all climate change stressors (e.g., ocean acidification, temperature) and invertebrate species in response to multiple stressors with and without modelled effects of ocean acidification (OA). In the upper panel, bold lines represent multi-model means while faint lines represent results from three ESMs (GFDL-ESM2G, IPSL-CM5A-MR, MPI-ESM-MR). Lines in the lower panel are multi-model averages of the three ESMs. Changes in MCP are 10-year running means and relative to the 2001–2010 average. In the lower panel, with and without OA effects refers to an explicit inclusion or exclusion of OA effects on growth and survival. Including those OA effects leads to a much higher percentage loss in invertebrate species MCP.

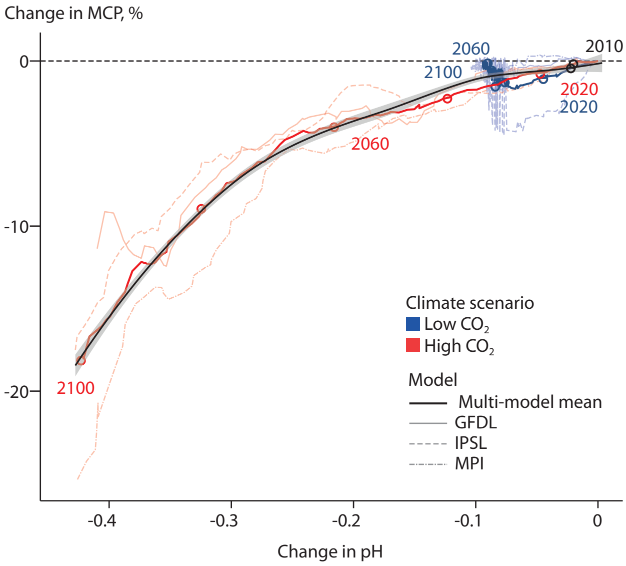


Figure A6.14 Projected changes in maximum catch potential (MCP) of marine invertebrate species due to ocean acidification in addition to other climate change stressors (e.g., temperature) for low and high climate change scenarios. Differences between model simulations with and without modeled ocean acidification impacts were calculated to determine how pH affects maximum catch potential. Faint colored lines represent simulations with the various ESMs and bold lines represent multi-model averages. Black line and grey bands represent the selected regression model and errors. Results shown are 10-year running means and relative to the 2001–2010 period. Open circles represent the year for the simulation results.

(Figure A6.15c,d). This is because, in this model, the northern regions are generally inhabited by very few, if any, species. Therefore, any invasions result in greater perceived turnover rates, as there were few species to begin with. The model shows that species invasions (and subsequently increases in MCP) in the Beaufort Sea area are restricted by the deep basin as many of the assessed species inhabit areas along the continental shelf (e.g., Majewski et al., 2017).

There is variability within species groups in response to ocean acidification impacts, even though the same parameterizations are used for each species group. However, mollusks showed the greatest sensitivity and variability to ocean acidification, while crustaceans showed negative but much more consistent effects. Clam species show the greatest variation in their response to ocean acidification, ranging from decreases of >25% to zero change (Figure A6.16). This suggests that there are multiple interacting factors with ocean acidification that work to amplify or attenuate impacts on species distribution, abundance, and MCP. Most species groups show a negative response to ocean acidification in the high CO_2 emission scenario, but some respond positively with increases in MCP. Nonetheless, results under the RCP8.5 scenario are far more variable with greater negative and positive responses to ocean acidification than results under RCP2.6.

Focus species: Arctic cod

The DBEM results for Arctic cod populations show largely negative responses to climate change. Under the high CO_2 emission scenario abundance decreases by more than 17% without the effects of ocean acidification (Figure A6.17). The projected effects of ocean acidification on abundance are small, amounting to an additional ~1% decrease in abundance. Arctic cod abundance shows significant decreases in the southern Arctic regions, and increases in northern regions, suggesting a northward shift for this species (Figure A6.18). However, the large decline in abundance of Arctic cod suggests that they may already be at their northern limits and thus their distribution range may be shrinking as it is compressed from the south (Frainer et al., 2017). The model suggests that ocean acidification has largely consistent negative effects across much of the species range (Figure A6.18).

The model includes the effects of ocean acidification on the life history traits: growth and survival. Results show that changes in abundance are more sensitive to effects on growth, and to a lesser extent on survival (Figure A6.19). Empirical evidence of the effects of ocean acidification on Arctic cod remains limited, but these results suggest that effects on survival may have fewer implications on species abundance than effects on growth. This also appears to be true for effects on the survival of Arctic cod larvae, where downstream effects on abundance are negligible.

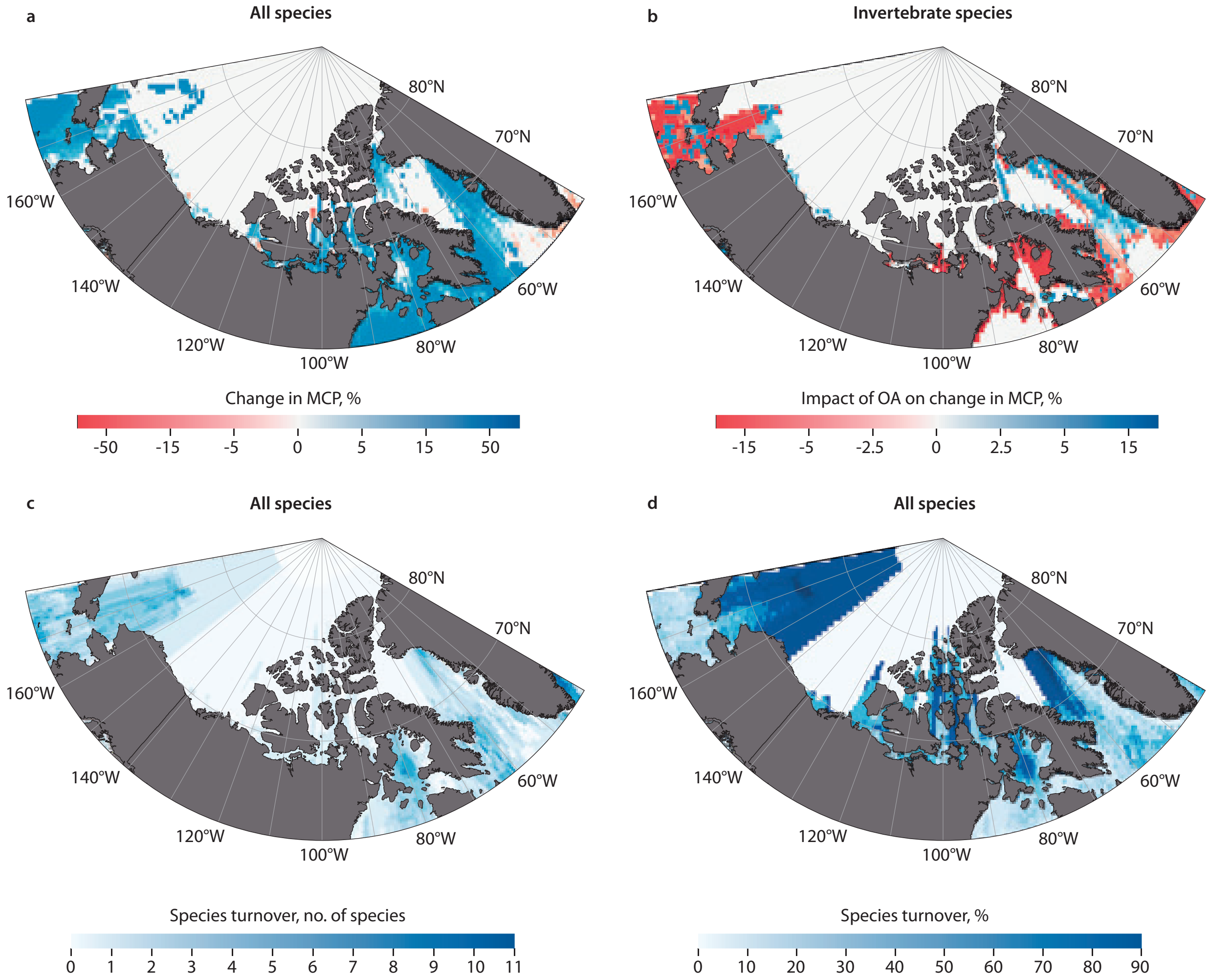


Figure A6.15 Projected species responses under the high CO_2 emission scenario (RCP8.5) by 2100 (relative to the 2001–2010 period) on the maximum catch potential (MCP) for all exploited species and the additional impact of ocean acidification (OA) for invertebrate species. The two lower plots show absolute species turnover number and the rate of species turnover.

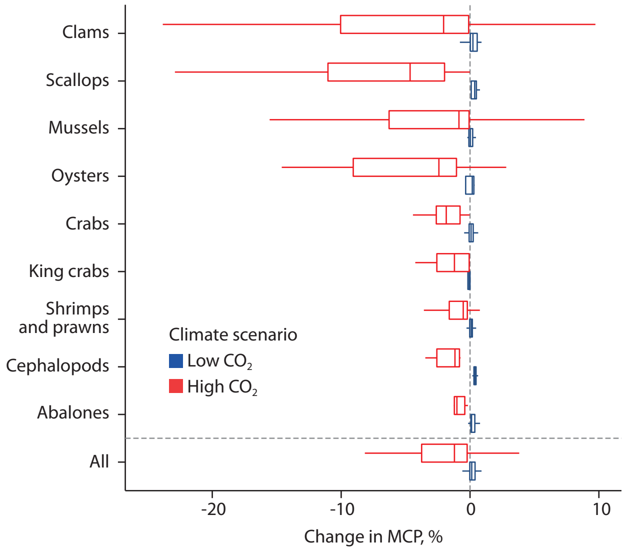


Figure A6.16 Ocean acidification impacts on maximum catch potential (MCP) by 2100, summarized by species group for two climate change scenarios: low CO_2 (RCP2.6) and high CO_2 (RCP8.5). Results are an average from 2091–2100 and are relative to the 2001–2010 period. The box plots use standard metrics. The middle line within the box is the median and the lower and upper hinges of the box are the first and third quartile (25th and 75th percentile) range. The extent of the whiskers goes to the furthest data point but no further than 1.5×IQR from the lower and upper hinges, where IQR is the interquartile range (distance between first and third quartiles).

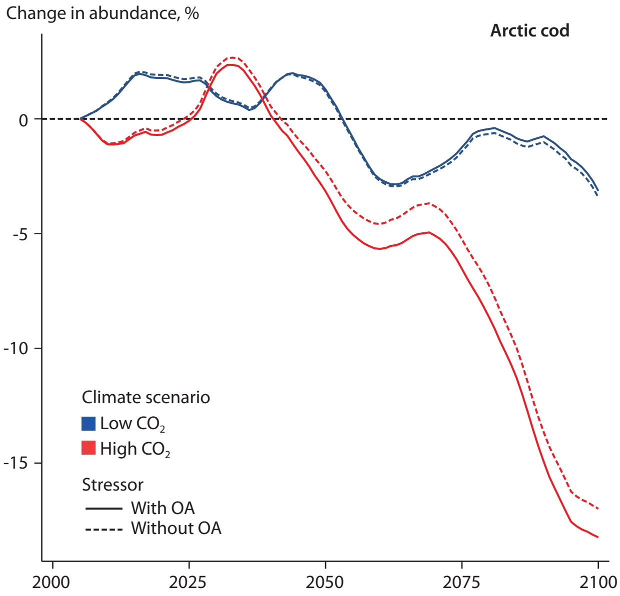


Figure A6.17 Projected changes in abundance for Arctic cod for models with and without the effects of ocean acidification. Each line is the multi-model average of the three ESMs (GFDL, IPSL, MPI). Changes in abundance are 10-year running means and relative to the 2001–2010 average. Red indicates the high CO_2 emission scenario (RCP8.5) and blue indicates low emissions (RCP2.6).

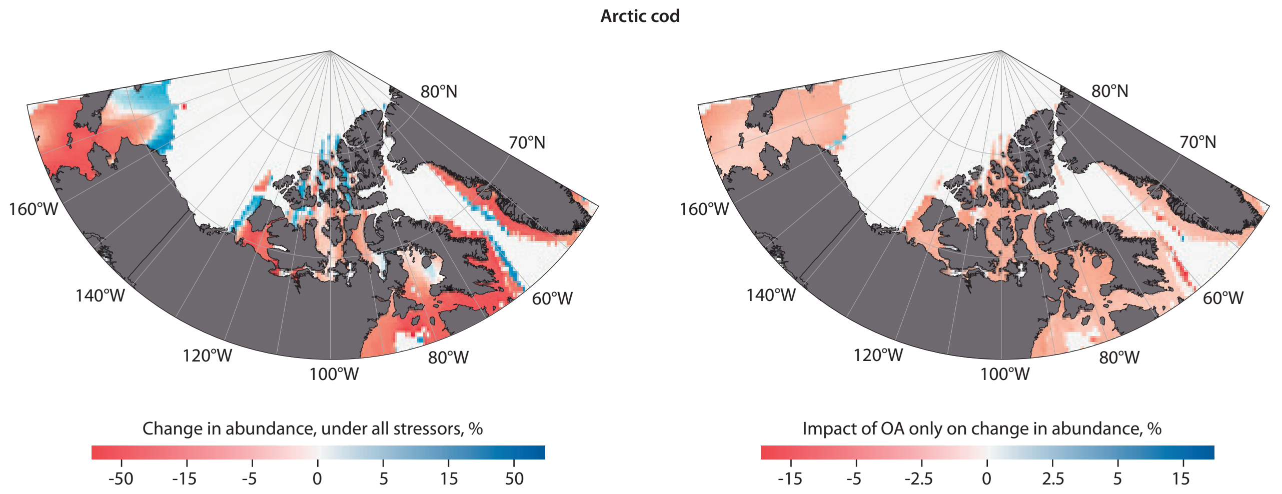


Figure A6.18 Projected changes in abundance for Arctic cod in response to all stressors (ocean acidification and other climate stressors, such as temperature) and in response to ocean acidification (OA) only. Changes in abundance are 10-year running average of multi-model means and relative to the 2001–2010 average.

Figure A6.19 Projected changes in abundance for Arctic cod due to ocean acidification (OA) using various values for modelled effect sizes of ocean acidification impacts: on both growth and survival (adult and larval), growth only, adult survival, and larval survival.

Potential fisheries impacts: economic model results

The economic model suggests that if commercial fisheries were to exist in the Canadian Arctic, current (2001–2010) sustainable catch would be over 55,000 tonnes for the species included in the analysis, with finfish accounting for almost 90% of fisheries production (Figure A6.20). With current global prices, Arctic fisheries would have a potential value of over USD 52 million (Figure A6.20). Due to the relatively higher price received, invertebrates accounted for 35% (USD ~18 million) of this total value.

DBEM projections show an increase in fisheries catch potential throughout the Arctic regions due to the poleward shift in distributions by the end of the century (2091–2100) (Figures A6.15 and A6.20). If Paris Agreement targets are met (low CO_2 emissions), modest increases could be expected in potential catch (10,500 tonnes) and landed value (USD 12.7 million) (Figure A6.20). In contrast, increases would be much larger with the high CO_2 emission scenario. Total potential catch was projected to increase by 107,000 tonnes to a total of ~163,000 tonnes per year. With current prices, this would equate to over USD 130 million annually, an increase of USD 69 million relative to current catch potential. Ocean acidification had small impacts on invertebrate fisheries catch under the high CO_2 emission scenario, reducing catch by ~900 tonnes valued at USD ~1.6 million.

An extraction for species-related estimates within the Beaufort Sea region is given in Table A6.2. The estimated values are directly related to projected biomass and do not take into account future changes in price. The table shows the large biomass contribution from forage species with significant increases in capelin and slight reductions in Arctic cod for the high CO_2 emission scenario, but also an increase in the subsistence species Arctic char. The current estimated value for Beaufort Sea fisheries is USD 4.5 million, and this is projected to rise in future scenarios. Under the low CO_2 emission scenario landed value is projected to show a marginal increase to USD 5.3 million, but under the high CO_2 emission scenario it is expected to increase to USD 15.7 million. This higher landed value is largely due to substantial increases in capelin, a relatively low value species. This explains the 5-fold increase in catch potential but only a 3-fold increase in landed value.

A6.5 Discussion

Model uncertainties

The issue of uncertainty in climate model projections was discussed in detail in the latest IPCC Assessment Report and focused research papers (e.g., Collins et al., 2013; Cubasch et al., 2013; Kirtman et al., 2013; Knutti and Sedláçek, 2013).

Uncertainty in climate model projections

Kirtman et al. (2013) described three main sources of uncertainty in climate projections (Figure A6.21). (1) Natural internal variability which is intrinsic to the climate system and places fundamental limits on the precision with which future climate variables can be projected. (2) Uncertainty

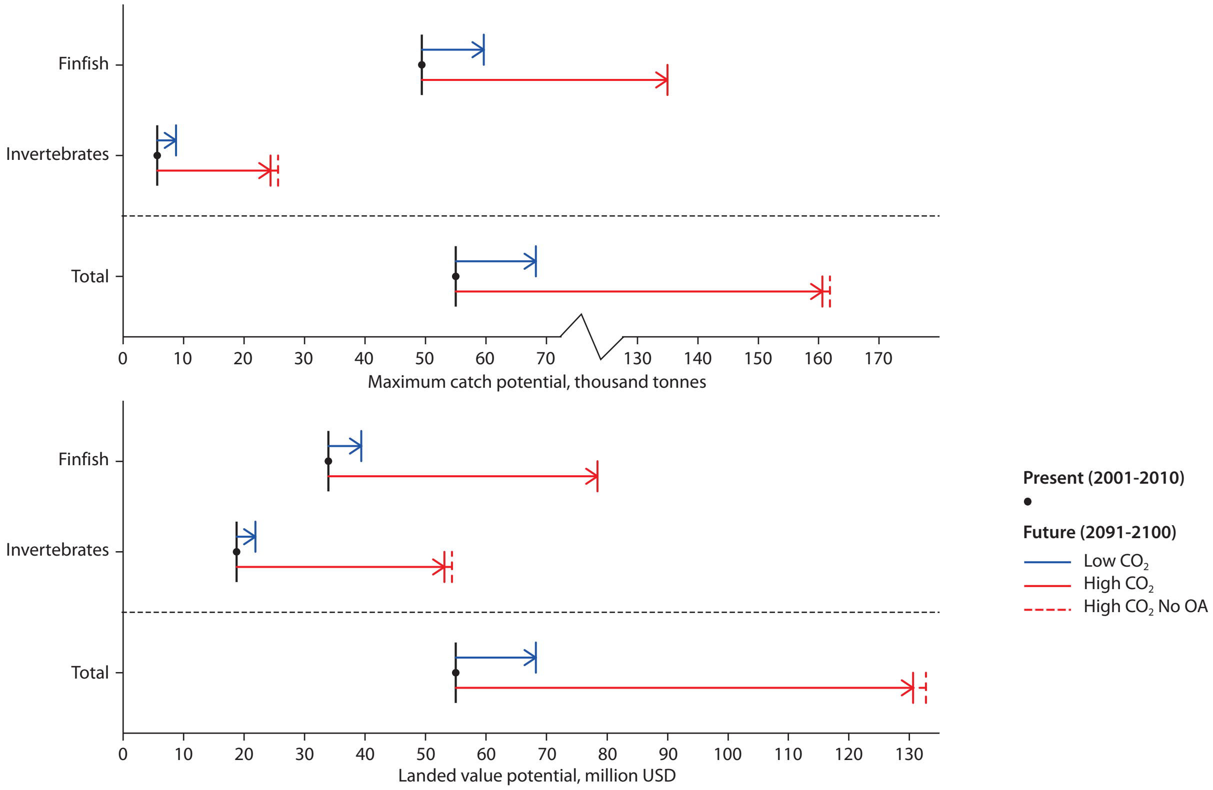


Figure A6.20 Estimates of present and future Arctic fisheries in Canada under a low and high CO_2 emission scenario. For invertebrate species only, solid lines represent model simulations with ocean acidification (OA) effects, while dashed lines represent model simulations without OA effects. Values presented are multi-model means of three ESMs and are 10-year running means.

Table A6.2 Estimates of catch (tonnes) and landed value (USD) of potential current fisheries and future projections under the low CO_2 emission (RCP2.6) and high CO_2 emission (RCP8.5) scenarios in the Beaufort Sea. Values are multi-model means from simulations using three ESMs (GFDL, IPSL, MPI) and values in parentheses are the minimum and maximum results from simulations. Average global ex-vessel prices from the 2001–2010 period were used to estimate landed values (Tai et al., 2017).

	Current (2001–2010)		RCP 2.6 (2091–2100)		RCP 8.5 (2091–2100)	
	Catch	Landed value	Catch	Landed value	Catch	Landed value
Arctic cod (*Boreogadus saida*)	4600	3240,000	4820	3400,000	4390	3090,000
	(2810 – 5800)	(1980,000 – 4090,000)	(3670 – 6610)	(2580,000 – 4660,000)	(2610 – 6510)	(1840,000 – 4590,000)
Capelin (*Mallotus villosus*)	4,310	1150,000	6230	1670,000	46,300	12,400,000
	(1260 – 6490)	(337,000 – 1740,000)	(3850 – 7810)	(1030,000 – 2090,000)	(41,900 – 48,800)	(11,200,000 – 13,000,000)
Navaga (*Eleginus nawaga*)	103	69,300	75.6	50,900	50.4	34,000
	(69.4 – 144)	(46,700 – 96,800)	(16.8 – 144)	(11,300 – 97,100)	(32.2 – 68.2)	(21,700 – 45,900)
Atlantic halibut (*Hippoglossus hippoglossus*)	10.5	77,700	9.31	68,800	14.0	104,000
	(1.90 – 16.4)	(14,100 -121,000)	(0.363 – 13.8)	(2680 – 102,000)	(0 – 25.2)	(0 – 186,000)
Lemon sole (*Microstomus kitt*)	9.42	45,500	10.8	52,100	15.8	76,300
	(6.81 – 11.3)	(32,900 – 54,500)	(9.24 – 12.6)	(44,700 – 60,700)	(14.6 – 17.8)	(70,800 – 86,100)
Arctic char (*Salvelinus alpinus alpinus*)	3.50	13,200	4.80	18,100	7.36	27,800
	(1.17 – 5.39)	(4410 -20,300)	(0.794 -9.47)	(2990 – 35,700)	(1.51 – 14.8)	(5690 – 55,700)
Greenland cod (*Gadus ogac*)	0.300	756	0.408	1040	0.392	1000
	(0 – 0.895)	(0 – 2290)	(0 – 1.22)	(0 – 3130)	(0 – 1.18)	(0 – 3010)
Total	9029	4599,000	11,100	5250,000	50,800	15,700,000
	(4150 – 12,500)	(2410,000 – 6120,000)	(7530 – 14,600)	(3670,000 – 7040,000)	(44,600 – 55,400)	(13,200,000 – 18,000,000)

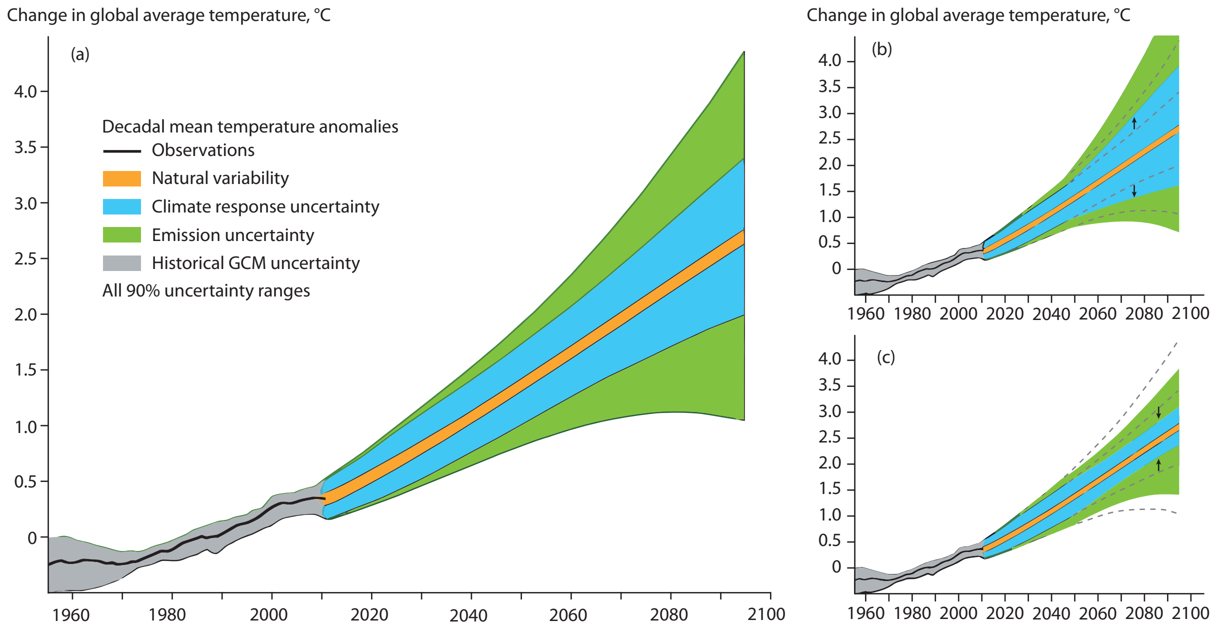


Figure A6.21 Relative importance of different uncertainties, and their evolution. The graphic shows (a) Decadal mean surface temperature change (°C) from the historical record, with climate model estimates of uncertainty for the historical period, along with future climate projections and uncertainty. Values are normalized by means from 1961 to 1980. Natural variability derives from model interannual variability, and is assumed constant with time. Emission uncertainty is estimated as the model mean difference in projections from different scenarios. Climate response uncertainty is based on climate model spread, along with added uncertainties from the carbon cycle, as well as rough estimates of additional uncertainty from poorly modelled processes. (b) Climate response uncertainty can appear to increase when a new process is discovered to be relevant, but such increases reflect a quantification of previously unmeasured uncertainty, or (c) can decrease with additional model improvements and observational constraints. The given uncertainty range of 90% means that the temperature is estimated to be in that range, with a probability of 90%. From Cubasch et al. (2013: FAQ 1.1 Figure 1).

concerning the past, present and future forcing of the climate system by natural and anthropogenic forcing agents such as greenhouse gases, aerosols, solar forcing and land use change (emission uncertainty). (3) Uncertainty related to the response of the climate system to the specified forcing agents (model uncertainty). The latter describes uncertainty about the extent to which any particular climate model provides an accurate representation of the real climate system and arises from approximations required in the development of models. Forcing-related uncertainty is estimated using the spread of projections for different emission scenarios (i.e., RCPs), while the spread among different models for individual scenarios is used as a measure of the model response uncertainty. However, due to their natural internal variability, ESMs have limitations in that they do not reliably simulate interannual and decadal variability, which might overlay and sometimes mask or enhance a longer-term trend (Taylor et al., 2012; Loder and van der Baaren, 2013; Swart et al., 2015).

Uncertainty in near-term projections is dominated by internal variability and model spread. This finding provides some of the rationale for considering near-term projections separately from long-term projections. Near-term projections are generally less sensitive to differences between future CO_2 emissions scenarios than long-term projections. Over short time scales evolution is largely dominated by the initial state while at longer time scales the influence of the initial conditions decreases and the importance of the forcing increases (Kirtman et al., 2013).

It is sometimes necessary to employ a dynamical or statistical downscaling procedure. Downscaling introduces an additional dimension of model uncertainty. The relative importance of the different sources of uncertainty depends on the variable of interest, the space and time scales involved (Meehl et al., 2007: their section 10.5.4.3) and the lead-time of the projection.

Uncertainty in the regional model

Regional-scale biogeochemical models of the Arctic are still under development and show inconsistent results with respect to the amount of primary production at current and future times (Popova et al., 2012; Vancoppenolle et al., 2013; Steiner et al., 2016). Differences are related to the opposing impacts of increased light versus decreased nutrient supply, and the representation thereof in the models, but also to the limited understanding of ecosystem functions such as the light-growth response in Arctic marine phytoplankton. Many biogeochemical processes are still insufficiently parameterized. Sea-ice algae which are closely linked to the diet of young Arctic cod have only recently been developed for regional Arctic models, and current models do not differentiate between sympagic and pelagic zooplankton species. Many processes which are responsible for mixing and nutrient supply are small-scale processes and require higher resolution modeling and adequate bathymetry information (e.g., coastal upwelling, wind mixing, tidal mixing). The Western Arctic Bioregion is still not well charted, with hydrographic surveys focusing on the main shipping routes.

No other higher resolution model projections of ocean acidification are currently available for the region, but Popova et al. (2014) did find high variability with respect to the progress of acidification and emergence of undersaturated waters in the Arctic in their model analysis.

Use of only one model under one forcing scenario implies high model and scenario uncertainty. The application of multiple models as well as additional scenarios would make it possible to estimate the range of uncertainty in the results.

Uncertainty in the DBEM

Scenario and model uncertainty with respect to the DBEM forcing is addressed via the use of three different models and use of a low (RCP2.6) and a high (RCP8.5) CO_2 emission scenario. With respect to uncertainties within the model, the DBEM projects changes in the distribution and abundance of each species on a 30' longitude by 30' latitude grid. As a result, outputs are best analyzed at larger geographic spatial units because the coarse resolution fails to capture coastal processes. Furthermore, the DBEM runs simulations on only one species at a time and so assumes that biological interactions and ecosystem effects are constant. Biological components could restrict or facilitate changes in distribution and abundance. In addition, the model only takes into account surface and bottom temperatures, whereas fish species inhabit a large range of depth intervals and might choose depths that are less prone to change and where they are less vulnerable.

While ocean acidification effects on species are highly variable, this analysis provides an overview of its potential impacts. The effects of ocean acidification on life history parameters are assumed to have a linear relationship, yet other mechanisms may be possible (Tai et al., 2018). For example, a threshold relationship would initially show minimal responses to ocean acidification but have exponentially greater impacts after a minimum threshold was crossed. Furthermore, the model assumes no potential acclimation/adaptation to ocean acidification or other climate stressors.

Uncertainties in the food web model

From the EwE (Ecopath with Ecosim) perspective, uncertainties are based on assumptions made to calculate or estimate parameters in the initial model, and changes over time. For the initial Ecopath model, input parameters were ranked based on how they were calculated using a pedigree analysis available in EwE (Christensen et al., 2007). Each input parameter was ranked based on the level of assumptions made when setting initial parameters (Figure A6.22). Overall, most is known about marine mammals including key parameters (biomass, production, consumption), as well as diet and changes over time. This knowledge is based on local studies that examine the populations in the modeled area. These groups have the highest confidence on population levels, changes, and role in the ecosystem. Much of the information known about links in the food web comes from research on higher trophic levels (stable isotopes and fatty acid analysis), giving insight into the structure and function of the lower trophic levels.

Fish groups have some of the highest uncertainty in terms of ecosystem research, as large-scale assessments on numbers of local species have only recently been collected under the Beaufort Regional Environmental Assessment (BREA) Marine Fishes program (2012–2015). Before this study, coastal programs were limited yet were the main source of biological parameters, numbers, and relative abundance of fish for the region. For these reasons, biological parameters were estimated

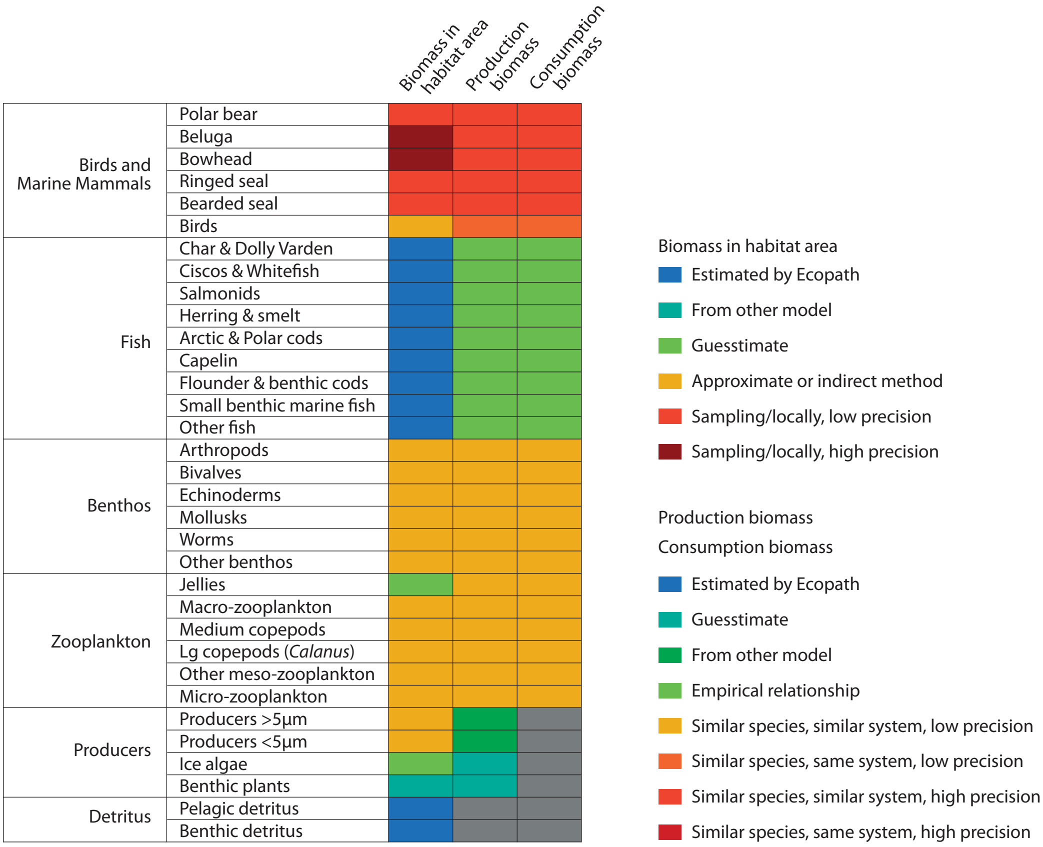


Figure A6.22 Uncertainty in initial EwE (Ecopath with Ecosim) model parameters based on pedigree analysis. Values range from high to low for each of the main key inputs: biomass, production (production/biomass), and consumption (consumption/biomass).

based on the same or similar species in other regions. Some large-scale benthic studies occurred in the 1970s, providing important information on abundance, distribution and key species (Wacasey, 1975; Wacasey et al., 1977). Zooplankton and production studies on abundance, distribution, and food-web role have taken place more recently, providing more relevant information on these groups. As the sampling of species diets occurs in the summer months due to logistics of working in the Arctic, there is a potential lack of understanding of annual diets from the food-web perspective, so switching from specialists to generalists adds uncertainty to individual species and ecosystem impacts.

While fish groups are the least certain in terms of the EwE model, this does not mean there is no knowledge of this group, but rather that it is based on information from other systems, species-specific studies (such as provided by the other sections of this report) and expert information. For future projections the added uncertainty of changes in catches will need to be addressed.

Uncertainty in the economic model

Estimates of the value of current and future fisheries are derived from current (2001–2010) average global prices of each species and weighted by catch tonnage. This assumes that prices received for landings in the Arctic would be comparable to the global market. Prices used to estimate future potential values of Arctic commercial fisheries were assumed to be the same as current prices. Therefore, this estimate does not account for supply-demand effects on price, nor does it account for inflation and estimated landed values in today's market. Landed values are gross revenues collected at the first point of sale; estimating profits would require estimates of the cost of fishing, including fuel, labor, and maintenance.

While the economic model estimates potential amounts and values of the fishing resources, the feasibility of fisheries in the Arctic depends on several other factors, including enhanced costs due to long access, limited capacity (shipping, processing), resource rights, governance and policies (see Section on *Law and governance context*).

Uncertainty in physiological responses for key marine species

Traditionally, critical limits were the focus of much early physiological research. However, sub-chronic limits to performance are potentially important as drivers of marine species distribution and more research is needed on these. Also required are studies that quantify the cumulative impacts of multi-stressors such as temperature, ocean acidification, noise and hypoxia.

Arctic cod have the ability to acclimate to different temperatures depending on the duration of exposure (Drost et al., 2016). This finding is not unexpected as it has been shown that even true stenothermal Antarctic fish species are able to acclimate to warmer temperatures (Pörtner et al., 2000; Lannig et al., 2005; Seebacher et al., 2005; Franklin et al., 2007; Robinson and Davidson, 2008; Peck et al., 2014). The challenge is quantifying acclimation potential as well as the cost and benefits of acclimation on ectotherms, both for individuals and populations. Results from this study and past research suggest that the cost of acclimation may, in some cases, outweigh the benefits (Woods and Harrison, 2001; Seebacher et al., 2005; Deutsch et al., 2015; Pershing et al., 2015; Drost et al., 2016).

One way to quantify acclimation potential is to evaluate the change in physiological performance curves due to change in conditions. For instance, aerobic performance curves for an aquatic animal can quantify thermal optimum windows based on peak values of aerobic scope, when the capacity to perform aerobically is at its highest. These curves are species- and even, in some instances, population-specific (Fry, 1947; Eliason et al., 2011). Pörtner and Farrell (2008) predicted that performance curves would shift to the right with warm acclimation.

Acclimation potential can also be quantified for a given species by contrasting the Q_{10} values at two different acclimation temperatures. The Q_{10} represents the factor by which the rate of a reaction increases over a specified rise in temperature (traditionally 10°C). When Q_{10} is 1 the rate of reaction has plateaued, that is, the reaction rate no longer changes with an increase in temperature. Therefore, when contrasting the heart rate of fish from two acclimation temperatures, a Q_{10} value of 1 would indicate complete acclimation. This application of Q_{10} was recently referred to as the Acclimation Potential Index – API (Seebacher et al., 2015). The API value for Arctic cod is 1.7.

There may be a significant cost to acclimation and the subsequent alterations in respiratory and cardiac performance (Pörtner and Farrell, 2008). Climate change multi-stressors may affect the ability of marine fish to acclimate. Recent experiments on cardiac mitochondria performance in Atlantic cod show that acclimation potential might be constrained by high partial pressure of carbon dioxide (pCO$_2$) (Leo et al., 2017). The potential to acclimate (e.g., thermal plasticity in fish) requires further study and this potential should be included in ecosystem models that project climate change impacts on food webs (Farrell and Franklin, 2016).

The physiological limits of 88 Arctic and subarctic marine species with potential relevance to the Western Arctic Bioregion have been identified to date (Steiner et al., 2018). Temperature thresholds are complete for 16 species, two species also have acclimation potential data and responses to ocean acidification are available for six species. While the database may not yet be complete, it clearly highlights knowledge gaps of species responses to environmental change. This causes high uncertainties in ecosystem response models which rely on species responses. Focused monitoring and experimental testing will reduce these uncertainties.

Potential impacts on subsistence fisheries

Empirical observations indicate a northward retreat of Arctic cod from their southern-most distributions, such as waters off Disko Bay, Greenland, Iceland-East Greenland waters, the Barents Sea and in the Inuvialuit Settlement Region (Hansen et al., 2012; Farrell et al., 2013; Astthorsson, 2015; Harwood et al., 2015). This observed northward shift in Arctic cod distribution would not have been predicted to have started this quickly based solely on their recently discovered cardio-respiratory thermal limits (Drost et al., 2014). The results from these thermal limits studies suggest that factors other than thermal tolerance and associated cardiac performance may influence the present-day distribution of Arctic cod. Young polar cod are strongly associated with the sea-ice habitat because ice-associated amphipods and copepods are its main prey. Biotic factors could include negative interactions with potential competitors such as capelin and at least two salmon species that are now migrating in greater numbers into the western Arctic (Dunmall et al., 2013; Suprenand et al., 2018). There is also evidence of a northward spread of Pacific zooplankton as far as the Beaufort Sea (Nelson et al., 2009, 2013) which might disrupt food supply for existing higher trophic level species.

The Ecopath model results suggest that a reduction in Arctic cod biomass is likely to have a negative impact on the Beaufort Sea marine ecosystem and would threaten walrus and ringed seal populations (Suprenand et al., 2018). This suggests a cascading impact could occur throughout the entire Arctic marine ecosystem. Analyzing body condition since the late 1980s in five marine vertebrate species, Harwood et al. (2015) identified ringed seal, beluga, and black guillemot (*Cepphus grylle*) chicks, all consumers with a dietary preference for Arctic cod, to experience declines in condition, growth and/or production. Over the same period they found increasing body condition in bowhead whale (subadults) and Arctic char, in both cases influenced by sea ice changes. A reduction in summer sea-ice extent, including increased duration of open water, changes in upwelling potential (wind stress) and possibly higher primary production (with shifts to higher small production; Li et al., 2009) in the marine ecosystem favor herbivorous zooplankton that are targeted by bowhead whales (e.g., Lowry et al., 1978; Walkusz et al., 2012; Nelson et al., 2013). The diet of Arctic char includes a substantial number of Arctic cod in some years and is dominated by zooplankton in other years (Figure A6.23). Harwood et al. (2015) stated that the proximate causes of the observed changes in body condition remain unknown, but may reflect an upward trend in secondary productivity (zooplankton), and a concurrent downward trend in the availability of forage fishes, such as the preferred Arctic cod. Among other factors, the body condition of marine vertebrates is directly linked to the total annual availability and quality of their prey, with nutritional stress ultimately linked to the health of individuals and populations (Moore and Gulland, 2014). The whale, seal and fish species analyzed by Harwood et al. (2015) are all important subsistence species in the Western Arctic Bioregion. Hence, a demise in Arctic cod populations could affect subsistence and commercial fisheries (Thorsteinson and Love, 2016). While changes in ice condition currently seem to favor zooplankton production, which could lead to an overall positive impact on subsistence fisheries, increasing Arctic ocean acidification and reductions in calcium carbonate saturation state might have negative effects on some Arctic zooplankton species, suggesting an uncertain future. A summary of responses of primary producers to ocean acidification is given in Chapter 3 and indicates variable responses from phytoplankton with overall fair resilience (Hoppe et al., 2017, 2018) and negative

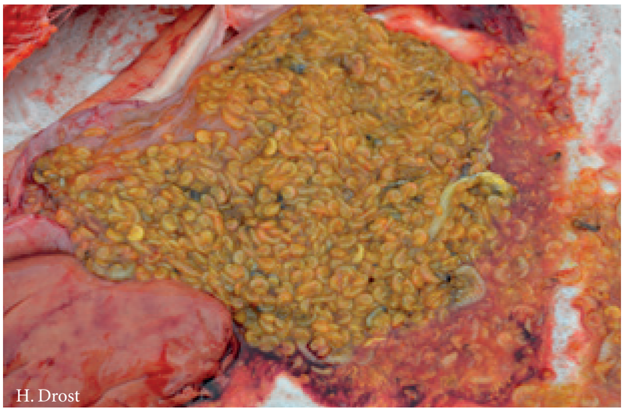


Figure A6.23 Arctic char – stomach contents containing about 40 Arctic cod (upper) (Cambridge Bay, 2012) and Arctic char, stomach contents dominated by zooplankton, Cambridge Bay 2011 (lower).

effects mostly for calcifying species (Yoshimura et al., 2013; Thoisen et al., 2015). Arctic pteropods are negatively impacted by temperature and ocean acidification (Comeau et al., 2010, 2012; Bednaršek et al., 2012; Lischka and Riebesell, 2012). For the Arctic copepod *Calanus glacialis* ocean acidification effects seem to vary with developmental stage. The developmental rate of nauplius larvae and the last copepodite stage appears largely unaffected whereas the earlier copepodite stages show increases in metabolic rate due to feeding at high $p\mathrm{CO}_2$ levels and decreased scope for growth (Thor et al., 2017). Such changes to earlier copepodite stages could have serious implications for the *C. glacialis* population, including prolonged stage development time and reductions in the individual body size of developing copepodites, adults, and ultimately the higher trophic level species feeding on them.

Monitoring the food chain

After analyzing body condition of several marine invertebrate species in the Western Arctic Bioregion, Harwood et al. (2015) highlighted the clear need for continued long-term, standardized monitoring of vertebrate body condition. The monitoring should be paired with concurrent direct (stomach contents) and indirect (isotopes, fatty acids) monitoring of diet, a detailed study of movements and seasonal ranges to establish and refine baselines, and the identification of critical habitats of the marine vertebrates being monitored. These studies aim at sampling the food chain and the factors controlling prey availability or quality and should be undertaken at temporal and spatial scales that are matched with, and relevant to, the marine vertebrate consumer. Continued food chain monitoring is also essential to track adaptation capacity of higher trophic levels to shifts in prey species and hence evaluate their sensitivity to environmental change.

Law and governance context

Laws and policies relevant to addressing the potential impacts of climate change and ocean acidification on marine species and coastal communities in the Western Arctic Bioregion appear at global, regional and national levels. The Paris Agreement continues to set the global agenda for advancing mitigation and adaptation responses to climate change and ocean acidification (Klein et al., 2017). Adaptation responsibilities set out in Article 7 of the Agreement (Pérez and Kallhauge, 2017) state the need for each Party to engage in adaptation planning processes and to implement adaptation actions based on a gender-responsive, participatory and fully transparent approach taking into consideration vulnerable groups, communities and ecosystems, and guided by the best available science and, as appropriate, the traditional knowledge of Indigenous peoples and local communities. This further highlights the need for a knowledge co-production approach when designing climate change and ocean acidification impacts studies.

Adaptation initiatives are also occurring at a regional level. These include the recent draft Agreement to Prevent Unregulated High Seas Fisheries in the Central Arctic Ocean (U.S. Department of State, 2017), the Arctic Council's Framework for a Pan-Arctic Network of Marine Protected Areas (PAME, 2015) and follow-up guidance on other area-based conservation measures (PAME, 2017) (see Chapter 1, Box 1.1).

Adaptation efforts have especially progressed at the national level. On the U.S. side of the Beaufort Sea, the North Pacific Fishery Management Council in 2009 approved a new Fishery Management Plan for Fish Resources of the Arctic Management Area. The Plan has closed Federal waters of the U.S. Arctic to commercial fishing until sufficient information is available to support the sustainable management of a commercial fishery. The Plan does not regulate subsistence fishing, recreational fishing or Alaska-managed fisheries (North Pacific Fishery Management Council, 2009). In 2016, the Alaska Ocean Observing System established the Alaska Ocean Acidification Network (www.aoos.org/alaska-ocean-acidification-network/about/mission), to engage with scientists and stakeholders to expand the understanding of ocean acidification processes and consequences in Alaska, as well as potential adaptation strategies.

At the Canadian national level, three initiatives have laid the foundation for implementing adaptive approaches to climate change and ocean acidification in the Arctic. First, pursuant to an *Oceans Act* mandate for the Minister of Fisheries and Oceans to lead the development and implementation of integrated management plans eventually covering all coastal and marine waters, the Beaufort Sea region was chosen as a pilot site for a large ocean management area (LOMA) integrated planning initiative. In 2009, the Beaufort Sea Planning Office published an Integrated Ocean Management Plan (IOMP) for the

Canadian Beaufort Sea, which includes the coastal and estuarine components of both the mainland and islands covering an area of 1750,000 km^2. The Plan was the collaborative work of people representing indigenous, territorial and Federal Governmental departments, management bodies, and northern coastal community residents with interests in the Beaufort Sea, and also includes input from industry and other interested parties. The Plan is voluntary and is intended to facilitate integrated planning among all Beaufort Sea resource users and managers, where shared responsibility for implementation of the Plan is intended to achieve responsible and sustainable use of the Beaufort Sea. The Plan has served as a support tool for MPA network planning in the region through the identification of EBSAs (Cobb et al., 2008).

Second, Canada is committed to achieving the MPA target adopted under the Convention on Biological Diversity of protecting 10% of marine and coastal areas by 2020. Two MPAs under the *Oceans Act* have been established in the Beaufort Sea region, the Tarium Niryutait MPA established in 2010 and the Anguniaqvia niqiqyuam MPA established in 2016. Fisheries and Oceans Canada in collaboration with Parks Canada Agency and Environment Canada is leading the design of an MPA network design for the Western Arctic Bioregion (www.beaufortseapartnership.ca/initiatives/western-arctic-bioregion-mpa-network). In June 2017, the government tabled Bill C-55 proposing to amend the *Oceans Act* in order to allow a faster process for establishing MPAs through a Ministerial Order as an interim step towards more formal MPA regulation within five years (DFO, 2017). Current marine conservation planning efforts in the Western Arctic Bioregion are aiming at including climate change and ocean acidification into their assessments by incorporating climate model downscaling to identify high risk areas (areas likely to experience high change under a changing climate), climate change refugia (areas likely to experience limited change under a changing climate), as well as changes to hotspots of biodiversity and primary production. The present case study is contributing to this effort.

Third, Canada has established a policy framework to ensure that future commercial fisheries in the Western Beaufort Sea Bioregion would be subject to precautionary and adaptive approaches. In 2014, Fisheries and Oceans Canada together with institutions established pursuant to the Inuvialuit Settlement Agreement (The Fisheries Joint Management Committee, Inuvialuit Game Council and Inuvialuit Regional Corporation) concluded a Beaufort Sea Integrated Fisheries Management Framework for the Inuvialuit Settlement Region (Fisheries and Oceans Canada et al., 2014). The Framework establishes a multi-step decision process for future commercial fisheries applications that would consider, among other factors, possible adverse effects on existing Inuvialuit subsistence fisheries and preferential rights of the Inuvialuit to harvest fish in the Inuvialuit Settlement Region (Ayles et al., 2016). In addition, future commercial fisheries would be subject to other DFO sustainable fisheries policies including those on the precautionary approach (DFO, 2009a) and on managing the impacts of fishing on sensitive benthic areas (DFO, 2009b).

Canada has advanced on various fronts to more broadly address climate change mitigation and adaptation. Canada ratified the Paris Agreement on 5 October 2016 and its 2017 Nationally Determined Contribution submission pledges a reduction in greenhouse gas emissions of 30% below 2005 levels by 2030. In December 2016, the Federal Government together with provinces and territories adopted the Pan-Canadian Framework on Clean Growth and Climate Change (Government of Canada et al., 2016). The Plan sets an agenda for pricing carbon pollution, reducing emissions across the economy, adapting to impacts of climate change and accelerating clean economic growth. Canada's three northern territories have adopted a Pan-Territorial Adaptation Strategy aimed at sharing experiences in adaptation approaches and challenges and developing collaborative activities (Governments of the Northwest Territories, Nunavut and Yukon, 2011).

A6.6 Summary and future directions

In this case study, a framework combining modelling and analysis tools has been introduced and applied to identify potential climate change and ocean acidification effects on Arctic marine ecosystems and impacts on subsistence fisheries in the Western Arctic Bioregion. Particular emphasis has been given to the key forage fish species Arctic cod (*Boreogadus saida*), since climate model projections can be linked more directly to key forage species than to the mostly higher trophic level species harvested. Changing environmental parameters include rising temperatures, a reduction in sea-ice thickness, area and seasonal coverage with related impacts on light available to the ocean, as well as changes in stratification and nutrient supply. These changes are suggesting an increase in primary producers in the Western Arctic Bioregion which has already been observed. Continued ocean acidification is indicated by an ongoing decrease in ocean pH and corresponding decrease in calcium carbonate saturation states for aragonite and calcite. Locations with aragonite undersaturation in surface and subsurface waters have already been recorded in Pacific inflow waters and within the Western Arctic Bioregion. Under a high CO_2 emission scenario (RCP8.5) complete aragonite saturation is projected for the surface and subsurface waters within 60 to 80 years. Waters are projected to remain supersaturated with respect to calcite over that same period but with significantly reduced saturation states. An emission scenario that includes mitigation will slow ocean acidification and can avoid calcite undersaturation but is unlikely to stop the current expansion of aragonite undersaturation.

Species shifts have been observed as well as modelled. Dynamic bioclimatic envelope model and food-web (EwE) model simulations suggest that northward migration constrains Arctic cod to the higher Arctic, but causes influx of other forage species such as capelin, sandlance (*Ammodytes hexapterus*) and zooplankton. The models show an increase in overall abundance and catch potential for the sum of species with a suggested continuation in the future (DBEM results available only) for most species and a decrease for some (e.g., Arctic cod, -17% by 2100). DBEM simulations which include potential responses to ocean acidification suggest variable responses within species groups. Mollusks showed the greatest sensitivity and variability to ocean acidification, while crustaceans showed negative but much more consistent effects. This suggests that there are multiple interacting factors with ocean acidification that work to amplify or attenuate impacts on species distribution, abundance, and catch potential. Most species groups show a negative response to ocean acidification under the high CO_2 emission scenario (RCP8.5), but some respond positively. Nonetheless, results for the RCP8.5 scenario are far more variable with greater negative and positive responses to ocean acidification than results for RCP2.6 (with mitigation according to the Paris Agreement). The results suggest that climate change and ocean acidification are likely to cause drastic changes to populations, resulting in large shifts in species composition potentially leading to changes in ecosystem structure. Observations suggest a decrease in body condition for higher trophic level species with a high preference for Arctic cod, including subsistence species such as beluga and ringed seal and increased body condition for species with more flexibility in food choices including bowhead whale and Arctic char. The overall impact on Arctic cod is projected to be negative for the high CO_2 emission scenario, but if subsistence species can adapt to alternative prey, the overall impact may be positive. The caveat is the still very limited knowledge of ocean acidification effects. For a low CO_2 emission scenario, in line with the Paris Agreement, both positive and negative impacts will be lower.

This initial assessment still includes significant uncertainties which need to be acknowledged and reduced to allow more precise projections. Scenario uncertainties in climate models and linked higher trophic level models can only be reduced with international commitments to reduce carbon emissions. Multi-model intercomparisons will make it possible to estimate a range for model parameter uncertainties, but those uncertainties can only be reduced with enhanced process understanding and parameterization development. In the Arctic, biogeochemical process models are still in their infancy and inter-model differences are large. Sea-ice algae which are closely linked to the diet of Arctic cod have only recently been incorporated into regional Arctic models. Many processes which are responsible for mixing and nutrient supply are small-scale processes and require higher resolution modelling and adequate bathymetry information (e.g., coastal upwelling, wind mixing, tidal mixing).

Large uncertainties in physiological species responses to individual and synergistic environmental changes can be reduced with enhanced monitoring and physiological testing on local species to better understand species response and improve predictive models. As a guidance for future directions the following key points emerge:

- Collaboration among disciplines studying environmental changes relevant to marine species, including physical, chemical, and other biological aspects of the environment are essential to improve understanding of the system and reduce uncertainties.
- Higher resolution oceanographic conditions, food web monitoring and comparative, multi-stressor physiological data of Arctic marine species are urgently needed to fully describe the potential climate change impacts on key food web Arctic and subarctic species.
- Improvements in model resolution, bathymetric information and model parameterizations based on enhanced process understanding are needed to improve predictive capacities.

A fragmented array of laws and policies has emerged relevant to addressing the potential impacts of climate change and ocean acidification on marine species and coastal communities in the Western Beaufort Bioregion. A brief discussion of global, regional and national dimensions leads to the following suggestions for future law and policy directions.

- While the need to strengthen marine management in general in the Arctic to counter the effects of ocean acidification on Arctic fisheries has already been emphasized (Sumaila et al., 2013), additional law and governance directions stand out.
- Ocean acidification threats and impacts need to be taken into account by all States and global society in progressively mitigating their carbon dioxide emissions as a matter of urgency (AMAP, 2013).
- The establishment of national MPA networks and a pan-Arctic MPA network need to be expedited as they are key paths to ensuring adaptation to climate change and ocean acidification.
- Future decisions regarding possible opening of commercial fisheries in the Western Arctic Bioregion and in the central Arctic Ocean high seas will need to be precautionary and adaptive in light of the many uncertainties still surrounding ocean acidification.
- Enhancing research and monitoring efforts in Arctic Ocean acidification must continue as a high priority within the Arctic Council (AMAP, 2013).
- Canada and the USA should operationalize and adequately fund collaboration on ocean acidification research and monitoring with the Western Arctic Bioregion being one of the priority areas for study.

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Acronyms and abbreviations

µatm	Micro atmosphere
Ω_{arg}	Saturation state of aragonite
Ω_{cal}	Saturation state of calcite
$CaCO_3$	Calcium carbonate
CH_4	Methane
CO_2	Carbon dioxide
CPUE	Catch per unit of effort
DBEM	Dynamic bioclimatic envelope model
DKK	Danish krone
ESM	Earth system model
EwE	Ecopath with Ecosim
fCO_2	Fugacity of carbon dioxide
ITQ	Individual transferable quota
MCP	Maximum catch potential
MSY	Maximum sustainable yield
NAFO	Northwest Atlantic Fisheries Organization
NOK	Norwegian krone
pCO_2	Partial pressure of carbon dioxide
RCP8.5	Business-as-usual emissions scenario
TAC	Total allowable catch
USD	U.S. dollar

Arctic Monitoring and Assessment Programme

The Arctic Monitoring and Assessment Programme (AMAP) was established in June 1991 by the eight Arctic countries (Canada, Denmark, Finland, Iceland, Norway, Russia, Sweden and the United States) to implement parts of the Arctic Environmental Protection Strategy (AEPS). AMAP is now one of six working groups of the Arctic Council, members of which include the eight Arctic countries, the six Arctic Council Permanent Participants (indigenous peoples' organizations), together with observing countries and organizations.

AMAP's objective is to provide 'reliable and sufficient information on the status of, and threats to, the Arctic environment, and to provide scientific advice on actions to be taken in order to support Arctic governments in their efforts to take remedial and preventive actions to reduce adverse effects of contaminants and climate change'.

AMAP produces, at regular intervals, assessment reports that address a range of Arctic pollution and climate change issues, including effects on health of Arctic human populations. These are presented to Arctic Council Ministers in 'State of the Arctic Environment' reports that form a basis for necessary steps to be taken to protect the Arctic and its inhabitants.

This report has been subject to a formal and comprehensive peer review process. The results and any views expressed in this series are the responsibility of those scientists and experts engaged in the preparation of the reports.

The AMAP Secretariat is located in Tromsø, Norway. For further information regarding AMAP or ordering of reports, please contact the AMAP Secretariat (The Fram Centre, Box 6606 Langnes, 9296 Tromsø, Norway) or visit the AMAP website at www.amap.no.

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