REVIEW / SYNTHÈSE

Climate change, fisheries, and aquaculture: trends and consequences for Canadian marine biodiversity[1]

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Abstract: Climate change, fishing, and aquaculture have affected and will continue to influence Canadian marine biodiversity, albeit at different spatial scales. The Arctic is notably affected by reduced quality and quantity of sea ice caused by global warming, and by concomitant and forecasted changes in ocean productivity, species ecology, and human activity. The Atlantic has been especially impacted by severe overfishing and human-induced alterations to food webs. Climate change, fishing, and aquaculture have all affected, to varying degrees, biodiversity on Canada's Pacific coast. Past and projected trends in key biodiversity stressors reveal marked change. Oceanographic trends include increasing surface water temperatures, reduced salinity, increased acidity, and, in some areas, reduced oxygen. Reductions in Canada's fishery catches (those in 2009 were half those of the late 1980s), followed by reductions in fishing pressure, are associated with dramatic changes in the species composition of commercial catches in the Atlantic (formerly groundfish, now predominantly invertebrates and pelagic fish) and the Pacific (formerly salmon, now predominantly groundfish). Aquaculture, dominated by the farming of Atlantic salmon, grew rapidly from the early 1980s until 2002 and has since stabilized. Climate change is forecast to affect marine biodiversity by shifting species distributions, changing species community composition, decoupling the timing of species' resource requirements and resource availability, and reducing habitat quality. Harvest-related reductions in fish abundance, many by 80% or more, coupled with fishing-induced changes to food webs, are impairing the capacity of species to recover or even persist. Open-sea aquaculture net pens affect biodiversity by (*i*) habitat alteration resulting from organic wastes, chemical inputs, and use of nonnative species; (*ii*) exchange of pathogens between farmed and wild species; and (*iii*) interbreeding between wild fish and farmed escapees. Physical and biological changes in the oceans, along with direct anthropogenic impacts, are modifying Canadian marine biodiversity with implications for food security and the social and economic well-being of coastal communities. To assess the consequences of changes in biodiversity for Canada's oceans and society, it is necessary to understand the current state of marine biodiversity and how it might be affected by projected changes in climate and human uses.

Key words: ocean, fish, overfishing, fish farming, Canada.

Résumé : Le changement climatique, la pêche et l'aquaculture ont affecté et continueront à influencer la biodiversité marine au Canada, bien qu'à des échelles spatiales différentes. L'Arctique se voit particulièrement affecté par des réductions qualitatives et quantitatives de la qualité des glaces marines causées par le réchauffement global ainsi que par les changements concomitants et prévus dans la productivité des océans ainsi que l'écologie des espèces et l'activité

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humaine. L'Atlantique a été particulièrement frappé par une surpêche grave et des modifications de la chaine alimentaire apportées par l'homme. Le changement climatique, la pêche et l'aquaculture ont affecté, à divers degrés, la biodiversité sur la côte pacifique du Canada. Les tendances passées et prévues chez les agents stressants pour la biodiversité révèlent d'importants changements. Les tendances océanographiques incluent l'augmentation des températures des eaux de surface, la réduction de la salinité, l'augmentation de l'acidité et, dans certaines régions, la réduction de l'oxygène. La réduction des prises dans les pêcheries canadiennes (réduction de 50 % entre 1980 et 2009), suivie d'une réduction de la pression des pêches, est associée à des changements drastiques dans la composition en espèces des prises commerciales dans l'Atlantique (auparavant des poissons de fond, aujourd'hui surtout des invertébrés et des poissons pélagiques) et dans le Pacifique (auparavant du saumon, maintenant surtout des poissons de fond). L'aquaculture, dominée par la production du saumon de l'Atlantique s'est développée rapidement entre 1980 et 2002 avant de se stabiliser depuis. On prédit que le changement climatique affectera la biodiversité marine en déplaçant la distribution des espèces, en modifiant la composition des communautés d'espèces, en découplant le moment des besoins en ressources des espèces avec la disponibilités des ressources et en réduisant la qualité des habitats. La réduction de l'abondance des poissons liée à la récolte, dans cas plusieurs de 80 % et plus, couplée avec les modifications induites par la les pêches dans la chaine alimentaire diminuent la capacité des espèces à reprendre ou même à persister. Les enceintes de culture implantées en mer affectent la biodiversité par : (*i*) l'altération de l'habitat provenant des déchets organiques, l'introduction de substances chimiques et l'utilisation d'espèces non indigènes; (*ii*) l'échange d'agents pathogènes entre les espèces d'élevage et les espèces indigènes; (*iii*) les croisements génétiques entre les poissons sauvages et les espèces d'élevage. Les modifications physiques et biologiques des océans accompagnées d'impacts anthropiques modifient la biodiversité marine au Canada avec des implications pour la sécurité alimentaire et le bien-être socio-économique des communautés côtières. Pour évaluer les conséquences des changements sur la biodiversité et la société, il faut comprendre l'état actuel de la biodiversité marine et comment elle peut être affectée par les changements projetés dans le climat et par les utilisations humaines.

Mots-clés : océan, poisson, surpêche, aquaculture, Canada.

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1. Introduction

"Nothing is so boundless as the sea, nothing so patient. It is not true that the sea is faithless, for it has never promised anything; without claim, without obligation, free, pure, and genuine beats the mighty heart, the last sound one in an ailing world. Many understand it scarce at all, but never two understand it in the same manner, for the sea has a distinct word for each one that sets himself face to face with it".

Kielland 1880

1.1. A case for conserving Canadian marine biodiversity

Canada's oceans constitute a vital biological, geochemical, and physical milieu that supports human health, societal well-being, and creation of wealth. Canada has the benefit of, and responsibility for, three marine coastlines that contribute to our society in numerous ways. For thousands of years, the oceans have provided habitat for species of traditional and cultural significance to aboriginal people. Today, sustainably exploited fish populations and environmentally responsible aquaculture operations should provide secure local and national access to the protein and oils contained in seafood, and Canada's oceans provide space for numerous recreational and commercial activities. On three sides of Canada, the physical integrity of natural coastlines reduces erosion and buffers the land from oceanic storms. Globally, marine life provides more than half the oxygen humans breathe and serves as a potentially rich source for modern pharmaceuticals.

The Millennium Ecosystem Assessment (an international scientific effort modelled after the Intergovernmental Panel on Climate Change (IPCC)) has unequivocally acknowledged the importance of aquatic and terrestrial biodiversity and ecosystems to human well-being and sustainable human development

(Millennium Ecosystem Assessment 2005). Indeed, there are compelling reasons to believe that reductions in Canadian and global marine biodiversity impair the ocean's capacity to provide a plethora of ecosystem services that contribute to the resilience of marine ecosystems and the well-being of humankind. From a strictly financial perspective, the case for sustaining marine biodiversity and protecting marine ecosystems is based on the argument that the importance of species can be determined by their marketable value (e.g., food, potential sources of medicine, recreational harvesting) and their ability to provide nonmarket goods and services (e.g., carbon sequestration, erosion control).

Although the monetary worth of Canadian marine ecosystems has yet to be quantified, their combined value may well be substantial, based on estimates for other ecosystems. For example, for the year 2002, the nonmarket ecological services provided by Canada's boreal forests have been valued at $703 billion (including $582 billion for the storage of carbon in forests and wetlands), more than ten times the net market value associated with the commercial extraction of wood (Pembina Institute 2009). Of course, many sectors of society would also argue that there are other good reasons for conserving biodiversity, including the potential for species to provide new drugs, foods, or genes; the role of biodiversity in contributing to people's enjoyment of the oceans, including educational, recreational, and inspirational experiences; and moral and ethical reasons for doing so (Kunin and Lawton 1996). Although these benefits can be difficult to quantify, they influence the ways in which society wishes to treat the marine environment.

1.2. Report overview

In November 2009, the Royal Society of Canada (RSC) convened an expert panel to advise on a series of questions related to the sustainability of Canada's marine biodiversity. The RSC selected a group of 10 people from Canada, the United Kingdom, and the United States who represented a wide range of scientific and policy-related expertise relevant to the questions posed in the panel's provisional terms of reference. These provisional terms of reference were reviewed, revised, and finalized at the first of the panel's three meetings in June 2010. The panel's report was released in February 2012.

The RSC report represents the only collation of information on marine life, oceanography, climate change, fisheries, and aquaculture in the context of Canada's national and international obligations to sustain marine biodiversity. The purposes of the report are

- to serve as an educational tool to increase awareness of Canada's oceans;
- to describe trends in Canada's oceans and marine biodiversity;
- to evaluate past, present, and forecasted changes in three stressors that affect marine biodiversity: climate change, fisheries, and aquaculture;
- to describe and forecast how these three stressors have affected, and are likely to affect, Canadian marine biodiversity;
- to determine whether Canada has fulfilled its commitments to sustain marine biodiversity;
- to provide broad, strategically based recommendations, each accompanied by key actions, to establish Canada as an international leader in oceans stewardship and marine conservation.

The report's primary audience can be described as interested members of the Canadian public, including Members of Parliament, decision-makers within the political and bureaucratic hierarchies of government, nongovernmental organizations, the natural and social scientific community, and industry.

Following this introductory section, section 2 provides information on the Panel's mandate (including clarification of issues addressed and not addressed by the Panel); marine biodiversity (what is it? why is it important to sustain?); greenhouse gas emissions; and panel procedures. Section 3 provides descriptions of Canada's physical and biological oceanography, including a "biological audit" of Canada's marine species diversity. The next three sections focus on trends in three stressors on biodiversity: climate change (section 4), fisheries (section 5), and aquaculture (section 6). The consequences of each of these stressors for biodiversity are presented in sections 7, 8, and 9.

Thus, the primary purpose of the present work is to provide a broad empirically based description of past and forecasted trends in three key stressors of, and their effects on, Canadian marine biodiversity. An accompanying paper (Hutchings et al. 2012a) begins by detailing Canada's obligations to sustain marine biodiversity at the international and national levels. Following this is an evaluation by the Panel of the extent to which Canada is fulfilling its commitments to sustain marine biodiversity. The Panel's overall conclusions and recommendations are described by Hutchings et al. (2012a).

2. The Expert Panel Report: mandate, content, and procedures

2.1. Terms of reference

Canada's long coastline and vast oceans give it a stewardship responsibility to be an international leader in addressing anthropogenic stressors that threaten ocean health and marine biodiversity. Climate change, fishing, and aquaculture influence marine biodiversity (albeit at different spatial scales) and pose challenges for managers and society. The Arctic Ocean is being affected by reductions in the quality and quantity of sea ice caused by global warming and concomitant changes in ocean productivity, ecology, and human activity. The Atlantic Ocean has been especially impacted by overfishing and associated changes in marine food webs. Climate change, fishing, and aquaculture are also affecting biodiversity on Canada's Pacific coast.

Physical and biological changes in these oceans, along with direct human impacts, can modify marine biodiversity with implications for food security and the social and economic well-being of coastal communities. To assess the consequences of changes in biodiversity for Canada's oceans and society, it is necessary to understand the current state of marine biodiversity and how it might be affected by projected changes in climate and human uses. Canada already has a range of national and international obligations that addresses aspects of marine biodiversity, but a key question is: Have Canada's actions been sufficient to sustain healthy, safe, and prosperous oceans for the benefit of current and future generations of Canadians?

The terms of reference identified the following questions for the Panel to consider during its deliberations:

1. What are the past and current trends and associated uncertainties in (*a*) physical and chemical indicators of climate change in Canada's three oceans and (*b*) Canadian marine biodiversity?
2. What are the projected consequences to Canadian marine biodiversity (and associated uncertainties) of climate change, fisheries, and aquaculture?
3. What are Canada's national and international obligations to sustain marine biodiversity, and to what extent are these obligations being fulfilled?
4. What new approaches and measures are required to promote the sustainability of Canadian marine biodiversity?
5. What research initiatives are required to support scientific advice to sustain Canadian marine biodiversity?

2.2. What is biodiversity?

The 1992 United Nations Conference on Environment and Development, Rio de Janeiro, dubbed the "Earth Summit", led to the Convention on Biological Diversity (CBD) and significantly increased national and international awareness of the conservation and sustainable use of biodiversity. The Convention defines biodiversity as "the variability among living organisms from all sources including, inter alia, terrestrial, marine, and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, among species and of ecosystems"; but it has been interpreted more broadly by society, policy makers, and scientists as the variety, quantity, and distribution of life (IEEP 2004). This is because quantity and distribution, like variety,

influence the function of ecosystems and the services they provide. Ecosystem services are the benefits that people derive from ecosystems, ranging from climate regulation to food production and recreation (e.g., Fisher et al. 2009).

The variety, quantity, and spatial distribution of living things describe variation among biological populations, species, or communities. Variety can be measured with metrics such as species richness (the number of species present in a defined area). The quantity of life (i.e., the abundance of plants and animals) determines organismal functions in ecosystems, the services they provide, and their chance of extinction. The distribution of life describes where populations, species, or communities are found in the ocean. This depends on environmental factors, such as climate, depth, and productivity, but also on abundance, since more abundant species typically have wider geographical ranges (e.g., Fisher and Frank 2004). The greater the variety, quantity, and distribution of life, the greater the biodiversity.

Biodiversity can be expressed at multiple levels. Unseen to the naked eye, there is variability within and among the genes of living organisms. Even though individuals of the same species — even within the same population — can look similar to one another, that similarity in "phenotype" (i.e., the visible characteristics of an organism) can mask variability at the genetic level. Genetic variability, thus, comprises one level of biodiversity. At another level, variability in phenotype (such as body colour, shape, behaviour, morphology, life history) can be considerable when one compares individuals from different populations. Population variability, then, comprises a second level of biodiversity. And there is, of course, tremendous variability at higher levels of biological organization, as reflected by the classification of organisms into various hierarchically arranged categories of taxonomy, such as species, genera, families, orders, and phyla.

Among the biodiversity indicators identified by the CBD are those that examine "trends in the abundance and distribution of selected species" and "trends in genetic diversity . . . of fish species of major socio-economic importance". However, at its core, the persistence of any species depends on the resistance and resilience of its component populations to anthropogenic and natural environmental perturbations (e.g., Schindler et al. 2010). This is the primary reason why the Panel focused on population trends in the Report (although species-level trends are presented where data exist). Secondary reasons include a lack of data on temporal trends in genetic diversity and the fact that an examination of trends in species numbers in Canada's oceans would not be particularly informative because few species have become extinct in our waters in recent millennia. Furthermore, there is ample precedent for using population trend data to describe temporal changes in marine biodiversity. For example, the population-level approach underpins the only marine index formally under consideration by the CBD — the Marine Living Planet Index (WWF 2010). Multispecies population indices have also been recently used to describe global trends in the biodiversity of marine fishes (e.g., Worm et al. 2009; Hutchings et al. 2010).

2.3. Why is biodiversity important to sustain?

International efforts to protect biodiversity are embodied by the objectives of the CBD to (*i*) conserve biological diversity,

(*ii*) use biological diversity in a sustainable fashion, and (*iii*) share the benefits of biological diversity fairly and equitably (United Nations 1992*a*). Reasons for the conservation and sustainable use of biodiversity range from its role in providing valuable or essential ecosystem services to a view that humans have moral and ethical responsibilities to "care" for life on earth (e.g., Kunin and Lawton 1996). Policy often focuses on service-related justifications, but sections of society are strongly influenced by moral and ethical concerns. Thus, where one person may see sustainability of resource extraction, another sees devastation of a fragile marine environment.

Diversity is directly related to persistence. The more variable things are, the more likely they will persist over time. Stock market portfolios typically reflect breadth to reduce the overall risk to one's investment capital. Farmers typically grow a variety of crops to reduce the chance of failure of any one particular crop. From a biological perspective, high genetic diversity increases the likelihood of having or producing genes that will allow adaptation to environmental change, including alterations to habitat or biological community brought about by natural variation and human actions. As well, the greater the genetic and phenotypic differentiation among populations, the greater the likelihood that some populations will be better able to respond favourably to environmental change than others.

The question often arises as to why biodiversity is important to sustain and conserve. Surely, one might ask, "Won't the biological functions of the oceans and their ability to provide renewable food resources be unaffected by the extinction of any one species or by the loss of any particular population of a species?" One response to this question is to draw an analogy between biodiversity and, say, the number of rivets that hold an airplane together. The loss of 1, 2 or possibly 10 rivets might not cause an airplane to fall apart. But if rivets continue to be lost, there will come to a point when the plane will not be able to function and catastrophic failure will ensue. The same is likely to be true for the functioning of marine ecosystems. The loss of 1, 2 or 10 species/populations might not be unduly problematic, but at some (unknown) cumulative biodiversity loss, catastrophic ecosystem change will ensue. Of course, one flaw with this analogy is the premise that all airplane rivets, and all biological species and populations, are equal in terms of their importance to the structural integrity of the plane or to the functioning of ecosystems. Our lack of knowledge of the functional importance of different species and populations, and of what their loss would mean to the functional integrity of marine ecosystems, has led to the adoption of a precautionary approach to the assessment, conservation, and protection of Canadian biodiversity as articulated, for example, by the *Species at Risk Act*.

2.4. Canada's stewardship responsibilities to marine biodiversity

A compelling argument can be made that Canada is an ocean nation. The country's motto, *A Mari Usque Ad Mare*, means "From Sea to Sea". The borders of eight provinces and three territories, comprising 86% of the Canadian population, are adjacent to salt water. At more than 200 000 km, it has been estimated that Canada has the longest coastline in the world and its oceans encompass an area (approximately 7 million km², DFO 2010*a*) roughly equivalent to 70% of

Canada's landmass and more than twice the size of India, the 7th largest country in the world. From a purely geographical perspective, the ocean stewardship responsibilities borne by Canada are arguably greater than those of any other country. Furthermore, given that most of Canada's coastline is located in Nunavut, the argument can be made that ocean issues in Canada are de facto Arctic issues, a region with which Canadians strongly identify.

The degree to which the health of Canada's oceans is a current priority for the federal government is unclear. The *Speech From the Throne* (3 June 2011) that opened Canada's 41st Parliament (Governor General of Canada 2011) made no reference to climate change, species recovery, fisheries rebuilding, or marine biodiversity. Neither the word "ocean" nor "Arctic" was mentioned in the throne speech. The "sea" is mentioned in the context of a government commitment to complete the Dempster Highway to connect Canada "by road from sea to sea to sea". And "fishing" is mentioned in the context of a government pledge to support it and other industries "as they innovate and grow". Asserting that the government has "expanded protected ... marine areas to an unprecedented extent", the throne speech states that "the Government will engage a broad range of stakeholders on the development of a National Conservation Plan", although details of this Plan, and the degree to which it will pertain to life in the oceans, are not indicated. For comparison, the preceding throne speech (Governor General of Canada 2008) did make reference to "tackling climate change and preserving Canada's environment", although oceans were not mentioned.

The near-absence of oceans issues in the throne speech could be interpreted as reflecting a lack of interest by the Government of Canada. It might also, however, be interpreted as reflecting a lack of interest on the part of Canadians. Perhaps the oceans are simply too distant — physically and experientially — for most people to feel strongly, one way or another, about the health of the marine environment.

However, rather than reflecting a real disinterest, any perceived disengagement of Canadians from the oceans might instead be attributable to the ways in which the oceans, and their relationship to Canadians, are communicated to society. It is not unusual, for example, for the oceans to be described primarily as a venue for human recreation, such as a "playground" for tourists (e.g., Explore Nova Scotia 2012) or an "adventure centre" for sport-fishing enthusiasts (Ocean Adventure Center 2011), rather than the primary global source of protein from wild animals. Fish and other commercially exploited marine organisms tend to be portrayed as commodities, rather than as integral biological components of ecosystems that comprise more than 70% of the planet's surface. Potentially confounding matters further, Fisheries and Oceans Canada (formerly Department of Fisheries and Oceans, and still widely known as DFO, which is the acronym that is used in this Report), the federal government department with primary jurisdictional responsibility for Canada's oceans, tends to identify its "clients", "partners", and "stakeholders" as members of the fishing industry (e.g., DFO 2008), rather than the Canadian public.

Yet, the oceans belong to no government and no industry. While all States enjoy various freedoms of the high seas, those freedoms are subject to numerous marine conservation responsibilities. Canada's oceans belong to the people of Canada, as the Supreme Court of Canada has affirmed (Supreme Court of Canada1997). In some countries, affirmation of societal ownership of the oceans is enshrined in legislation. Norway's *Marine Resources Act* (2009), for example, establishes the principle that the rights to wild marine resources belong to Norwegians; and it sets out clearly the state's responsibility to manage marine resources for the common good, acknowledging that "wild living marine resources belong to Norwegian society as a whole" (section 2; *Marine Resources Act*). By contrast, the preamble to Canada's *Oceans Act* is more circumspect, acknowledging that the oceans "are the common heritage of all Canadians" and that "the oceans and their resources offer significant economic opportunities for economic diversification and the generation of wealth for the benefit of all Canadians, and in particular for coastal communities."

Given this recognition of public interests, stewardship of Canada's oceans is a national responsibility of all Canadians. From a governmental perspective, the DFO's clients, stakeholders, and partners comprise all of the people of Canada, not simply those who obtain direct financial benefits from the extraction of marine resources. This stewardship carries with it the burden and responsibility of international leadership in the protection, conservation, and sustainable exploitation of marine biodiversity.

It is intended that this Expert Panel Report will assist decision-makers and Canadian society in their joint assessment of the degree to which Canada has embraced its national and international ocean stewardship responsibilities to protect, conserve, and sustainably exploit marine biodiversity. Furthermore, we hope that this Report's recommendations will serve to strengthen efforts to meet those responsibilities and allow Canada to be *the* world leader in ocean stewardship.

2.5. Scientific and extra-scientific issues

2.5.1. Drivers of change in marine biodiversity

Climate change, fisheries, and aquaculture are among the anthropogenic activities known or hypothesized to negatively affect organisms in Canada's oceans. Thus, these drivers of biodiversity change merit attention. An additional consideration is that the biodiversity impacts of these drivers are manifest at a range of spatial scales. The effects of climate change on biodiversity have been, and are forecast to be, significant in all three of Canada's oceans. The predominant factor affecting recent past and present trends in the abundance of most marine species has been fishing; this driver of change has been (to date) of greater importance in the Atlantic and Pacific, rather than the Canadian Arctic. The biodiversity impacts of aquaculture, because of potential transfer of disease from farmed to wild species, might be manifest across broader spatial scales than the relatively small areal extent of farms in Canada's marine environment. Although other variables are likely to affect marine biodiversity, their spatial scales of influence are either small relative to those associated with climate change and fisheries (e.g., coastal pollution and shipping, although the latter is likely to be of increasing importance in the Arctic with the loss of sea ice) or little studied in

Canadian waters (e.g., invasive marine species). A previous RSC Expert Panel Report addressed issues related to oil and gas exploration activities in coastal British Columbia (RSC 2004).

2.5.2. Clarification of issues addressed and not addressed by the Panel

The breadth of the Panel's terms of reference necessitated the imposition of limits on what the Panel could address in this Report. The Panel acknowledges, for example, that societal discussions, evaluations, and debates concerning the potential effects of climate change, fisheries, and aquaculture on biodiversity are not entirely scientific. Here, the Panel echoes observations made by another RSC expert panel, one that examined the potential risks to human health, animal health, and the environment associated with the development, production, and use of foods derived from biotechnology (RSC 2001).

Specifically, the Panel accepts that debates pertaining to conservation, climate change, fisheries, and aquaculture often fall into the following three kinds of discord. The first are *scientific* disagreements about what constitutes sustainability from a biodiversity or exploitation perspective. Among other issues, these debates can centre on questions related to target levels of population abundance and harvesting pressure, merits of alternative forecasting methods, and scientific uncertainty. The second form of disagreement can be described as *political* disagreements about the social and economic impacts of biodiversity loss, climate change, fisheries, and aquaculture. (Here, the term "political" refers to opinions relating to, affecting, or acting in accordance with the interests of status or authority within an organization rather than matters of principle.) Finally, there can also be *religious, ethical, and philosophical* disagreements about topics encompassed by the Panel's terms of reference, such as the question of whether *any* level of biodiversity loss or anthropogenic modification of the marine environment is acceptable.

Despite the existence and merits of these alternative forms of disagreement, debate, and discussion, this Report will focus primarily on the scientific elements of the questions posed in the terms of reference. More so, it will do so from a natural-sciences perspective rather than a social-sciences perspective. One justification for doing so lies in the Panel's conclusion that the social and economic impacts of biodiversity loss, climate change, fisheries, and aquaculture are sufficiently broad and multifaceted to warrant a separate comprehensive assessment. A second reason for restricting the Panel's efforts in this way lies in the contention that social, economic, political, and ethical discussions of the matters at hand are best preceded, and informed by, empirical documentation and scientific assessment of the potential biological consequences of climate change, fisheries, and aquaculture on Canada's ability to sustain marine biodiversity.

As additional points of clarification, the substantive sections of this Report do not explicitly address issues that were outside of either the Panel's mandate or its terms of reference. Excluded topics include Canada's role in international climate change negotiations and greenhouse-gas emission targets; government policies and legislation pertaining to the mitigation of greenhouse-gas emissions; potential biodiversity impacts of oil and gas development, pollution, coastal development, and shipping; aboriginal governance; or the adequacy

or appropriateness of particular fisheries and aquaculture management tools, such as individual transferable quotas, effort- versus catch-based controls on fishing effort, and fishery-license buy-outs.

From a terminological perspective, the Panel uses the term "fisheries" to refer to all past and present forms of extraction of marine organisms, such as fishes, mammals, invertebrates, and plants, and including those species that are caught incidentally by fishing gear, i.e., by-catch. When considering fishes, the Panel considers both "marine" and "diadromous" fishes. The former spend their entire lives within the ocean (e.g., Atlantic cod, *Gadus morhua*), whereas the latter typically spend part of their lives in freshwater rivers or lakes and part of their lives at sea (e.g., Atlantic salmon, *Salmo salar*; white sturgeon, *Acipenser transmontanus*; striped bass, *Morone saxatilis*; eulachon, *Thaleichthys pacificus*).

2.5.3. Recovery of marine biodiversity: societal influences

The degree to which the marine biodiversity losses documented in the Report are reversible will depend a very great deal on the wishes of society. Some reduction in fish abundance, for example, is unavoidable if society wishes to harvest fish for food. There are and always will be trade-offs associated with human use of the oceans and human protection of the oceans. But it is not the purview of a scientific panel to judge the relative importance of the societal benefits and costs associated with various marine-related activities.

Science can, however, provide advice on the degree to which biodiversity losses are reversible, if society asserts that such recovery is desirable. Many of the biodiversity consequences of aquaculture, for example, are likely to be readily reversible. The farming of fish in closed-containment facilities in water or on land would considerably reduce the environmental footprint of aquaculture. Affected marine ecosystems would likely recover within years, rather than decades, of such changes to fish-farming practices. The reversibility of some aquaculture-related biodiversity concerns, however, such as disease transfer and interbreeding with members of the same species, might be more problematic (section 9). As for fisheries, many overexploited populations and species are likely to increase in abundance if fishing pressure is reduced. However, the magnitude of recovery likely depends on multiple factors (e.g., fishing-induced changes to species life histories and marine ecosystems). And, even if fisheries are closed, the recovery of depleted fish populations — which is not guaranteed — is likely to require decades (section 8).

Climate change, on the other hand, is likely to be associated with a far greater degree of "permanence" in its effects on biodiversity and lower probabilities of biodiversity recovery than either fisheries or aquaculture. Unlike fisheries and aquaculture, both of which are regulated by Canadian jurisdictions, the magnitude and rate of climate change are outside of Canada's direct control, insofar as Canada is responsible for a relatively small proportion of the global greenhouse gas emissions (although it is among the highest per capita emitters). But this lack of direct control over the magnitude of global emissions does not absolve Canada of its responsibilities to protect marine (and terrestrial) ecosystems from the effects of climate change. In this regard, a brief description of Canadian and global

emission targets is appropriate, given the contributions of these emissions to climate change.

Canada, the United States (US), and 190 other countries are parties to the United Nations Framework Convention on Climate Change (United Nations 1992*b*, p.4), whose objective is

> ". . . stabilization of greenhouse gas concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system. Such a level should be achieved within a time frame sufficient to allow ecosystems to adapt naturally to climate change, to ensure that food production is not threatened and to enable economic development to proceed in a sustainable manner."

While the UNFCCC does not define what "dangerous" means, it does provide some guidance insofar as the treaty states that steps should be taken to ensure that (*i*) ecosystems can adapt naturally; (*ii*) food production is not threatened; and (*iii*) economic development can proceed in a sustainable manner.

The Kyoto Protocol to the UNFCCC, signed by both Canada and the US, was adopted in 1997. Annex B (developed) countries agreed to reduce greenhouse gas emissions to 5% below 1990 levels on average by 2008–2012. Canadian and American targets were 6% and 7%, respectively, below 1990 levels. Canada ratified the Protocol on 17 December 2002; it was never ratified by the US. By 2008, Canada's emissions were 24% above 1990 levels; US emissions had increased by 14%. By comparison, emissions in the United Kingdom had decreased by 18% and in Germany by 22%. The Canadian Government eventually announced that it would not try to meet its Kyoto target. Instead, in 2007, a "Made-in-Canada" solution was proposed: Canada would reduce its emissions to 3% below 1990 levels by 2020. Canada officially withdrew from the Kyoto Protocol on 12 December 2011.

At the December 2009 15th Conference of Parties (COP) to the UNFCCC, the nonbinding Copenhagen Accord was put together as a potential follow-up to the Kyoto Protocol. A total of 114 countries have "taken note" of the Copenhagen Accord which states that

> "deep cuts in global emissions are required . . . with a view to reduce global emissions so as to hold the increase in global temperature below 2 degrees Celsius."

In essence, the Copenhagen Accord revealed a broad international consensus, defining the use of *dangerous* under the UNFCCC: Global warming should be kept to less than a 2.0 °C increase from preindustrial times (this constitutes a further warming of 1.3 °C when compared with present-day temperature).

Shortly thereafter, Canada changed its position again, announcing that it was going to match the US number proposed by President Obama as part of the Copenhagen Accord. Obama promised that, by 2020, the US would reduce emissions by 17% relative to 2005 levels. This is equivalent to saying that, by 2020, US emissions would decline 3.6% relative to the 1990 level. In Canada's case, emissions would actually rise 2.5% from 1990 levels by 2020.

In the last few years, there have been significant advances in our understanding of the carbon cycle. It is now widely recognized that the lifetime of anthropogenic atmospheric CO_2 is very long (Eby et al. 2009; Solomon et al. 2009). Montenegro et al. (2007), for example, showed that if existing fossil fuel reserves were burnt, about 75% of the resulting CO_2 would remain in the atmosphere for 1800 years; 25% would remain for more than 5000 years.

An unfortunate corollary of this increased understanding is the recognition that stabilization of atmospheric CO_2 at any level requires anthropogenic CO_2 emissions to eventually decline to zero. However, the actual level of stabilization is independent of the emissions pathway. Rather, stabilization is determined by the cumulative emissions of anthropogenic CO_2 (Allen et al. 2009; Matthews et al. 2009; Meinshausen et al. 2009; Zickfeld et al. 2009). From a policy perspective, these realities have certain advantages. If a future level of atmospheric CO_2 stabilization or, equivalently, a future maximum allowable temperature increase is targeted, allowable future cumulative emissions can then be calculated.

Two recent analyses have estimated allowable cumulative emissions for limiting global warming to less than 2 °C from preindustrial values. In the first analysis (Zickfeld et al. 2009), it was assumed that any increase in human-produced, noncarbon dioxide greenhouse gases would be balanced by an increase in sulfate aerosols (or some other negative radiative forcing). While fairly accurate at present, this assumption should be viewed as extremely conservative, given that most future emission scenarios incorporate decreasing sulfate emissions and increasing emissions of noncarbon dioxide greenhouse gases. Zickfeld et al.'s (2009) best estimate of allowable cumulative emissions post-2000 (with a 66% probability of keeping warming below 2 °C) was 590 gigatonnes of carbon (GtC). In the second study (Meinshausen et al. 2009), a reduction in the allowable emissions budget was computed, using various scenarios for aerosols and non-CO_2 greenhouse gases (Nakicenovic et al. 2000). The researchers found that limiting CO_2 emissions to 390 GtC from 2000 to 2050 led to a 50% chance of exceeding 2 °C by 2100. Considering that about 92 GtC have been emitted since 2000 and about 10 GtC (and rising) are emitted annually, it is unlikely that temperatures will be kept below 2 °C without rather dramatic emissions reductions in the near future.

The disconnect between scientific estimates of allowable emissions and the voluntary targets communicated by nations to the secretariat of the UNFCCC is rather profound. Recent analysis (Rogelj et al. 2010) has shown that, even if countries meet their commitments under the Copenhagen Accord, the 2 °C warming threshold will almost certainly be crossed. This work further reveals a 50–50 chance that preindustrial temperatures will be exceeded by 3 °C during the 21st century. If emissions are not curtailed, the temperatures will increase by more than 4 °C relative to preindustrial levels.

Clearly, management actions to reduce the rate of climate change, and to control unsustainable but direct pressures on the marine environment, will affect the state of biodiversity on different time scales. Whereas actions to reduce CO_2 emissions and stabilize global climate will ultimately help to reduce rates of biodiversity change, these actions are not essential to meeting many existing objectives for the conservation and sustainable use of marine biodiversity. Importantly, by achieving existing objectives for biodiversity conservation and protection, Canada and other countries might help to mitigate further and unwanted ecological effects of climate change, thereby gaining additional time to

reach the international agreement targets required to reduce global CO_2 emissions.

3. Canada's oceans

3.1. Introduction

Canada's marine waters encompass an estimated 7.1 million km² (DFO 2010*a*). This oceanic area can be partitioned into different sections of sea, based on distances extending from the country's "baseline". As reported by DFO, the normal baseline is the low-water line along the coast, islands, rocks, including low-tide elevations, as marked on large-scale charts officially recognized by Canada. The length of highly irregular coastlines, such as that of Canada's, can be estimated by drawing straight baselines joining "appropriate points on the coast" (DFO 2010*a*). Based on such a procedure, Canada's coastline has been estimated at approximately 240,000 km, the world's longest.

Of the 7.1 million km² of ocean, the largest portion (2.9 million km²) is that contained within Canada's Exclusive Economic Zone (EEZ; extending 12 nautical miles from the coastal baseline to 200 nautical miles). Canada's internal waters (all waters landward of a coastal state's jurisdictional coastline) comprise 2.5 million km², and an additional 0.2 million km² make up Canada's territorial sea (0–12 nautical miles from the baseline). The waters overlying Canada's continental shelf beyond the EEZ (comprising the seabed and subsoil of the submarine areas that extend beyond the territorial sea throughout the natural extension of Canada's land territory to the outer edge of the continental margin) is currently estimated to be 1.5 million km². Canada has until 2013 to support this estimate with scientific and legal information as part of its eventual submission to the Commission on the Limits of the Continental Shelf, a UN body. Canada's extended continental shelves are situated on the Atlantic and Arctic coasts. The Commission's decision regarding Canada's submission will have implications for biodiversity, given that species in the benthos (sea bottom) on the extended continental shelves that are considered "sedentary" will be under the jurisdictional control of the adjacent coastal state.

Canadian marine biodiversity is ultimately a function of the physical and biological oceanography. The primary objectives of this section are threefold. The first is to present basic information on the geography, currents, circulation patterns, and water masses of those parts of the Atlantic, Arctic, and Pacific oceans that are adjacent to Canada's coast. The second objective is to provide basic information on the biological oceanography of Canada's waters, focusing on energy transfer and a depth-based delineation of biological ecosystems, leading to a description of the Canada's marine ecoregions. Thirdly, this section presents an overview of Canadian marine biodiversity from a species-level perspective.

3.2. The northwest Atlantic Ocean

3.2.1. Geography

The Atlantic Ocean, the world's second largest (approximately 82 million km²), is connected to the Arctic Ocean via Fram Strait (situated between northeast Greenland and Svalbard) and the Barents Sea. Its average depth (3900 m) is approximately double the deepest average depth in Canadian waters (1900 m in the Labrador Sea; Fig. 1).

The Greenland–Scotland Ridge, with maximum depths between 600 and 800 m, separates the Greenland, Iceland, and Norwegian seas from the rest of the North Atlantic. Deep waters originating in these Nordic Seas flow into the North Atlantic primarily via two deep-water pathways: Denmark Strait (620 m deep sill; a sill is a submerged ridge at relatively shallow depth separating the basins of two bodies of water) and Faroe Bank Channel (840 m deep). Periodically, water also overflows the shallow Iceland–Faroe and Wyville–Thomson ridges.

In Canada, Hudson Bay is an inland sea with a surface area of 819 000 km² and an average depth of ~100 m. It is the ultimate sink for 30% of Canada's freshwater runoff. Water leaving Hudson Bay joins the northwest Atlantic Ocean via Hudson Strait with an average depth of 275 m. To the north, Foxe Channel connects Hudson Bay to the shallow Foxe Basin (Fig. 1). Within Canada, the other major freshwater source to the northwest Atlantic is the St. Lawrence River (mean annual flow rate of 12 300 m³ s⁻¹ at Québec City).

The continental shelves along the east coast of Canada are typically very wide. Off Nova Scotia, the Scotian Shelf extends offshore up to 230 km with an average depth of 90 m. To the north, the Grand Banks comprise Canada's widest continental shelf, extending nearly 480 km. Most of the shelf is less than 150 m deep; some areas are as shallow as 25 m. Along the coast of Newfoundland and Labrador, the continental shelf has an average extent of about 150 km. The provinces of Nova Scotia, Newfoundland and Labrador, New Brunswick, and Prince Edward Island have a combined estimated total coastline length of approximately 40 000 km.

3.2.2. Circulation and water masses

Deep-water formation in the North Atlantic occurs in the Greenland Sea. The renewal of the deep North Atlantic is actually fed by an overflow of intermediate-depth water from the Nordic seas (Aagaard et al. 1985). About 1 Sv (1Sv = 10⁶ m³ s⁻¹) of northern source water passes between Iceland and the Faroe Islands and ~2 Sv passes between the Faroe Islands and Scotland (Østerhus et al. 2008). As these overflow waters flow southwestward into a deep western boundary current, they entrain surrounding waters, yielding ~3.2 Sv of transport southeast of Iceland (Saunders 1996; Hansen and Østerhus 2000). While some of the resulting overflow waters recirculate around the deep Iceland basin, ~2.4 Sv passes through the Charlie Gibbs Fracture Zone (Saunders 1994), eventually heading northward into the Irminger Sea (Fig. 2).

In addition, a nearly equal volume (~4 Sv; Dickson et al. 2008) of slightly colder northern source water passes over the shallow sill in the Denmark Strait, rapidly entraining surrounding water (Price and Baringer 1994) and yielding about 5.2 Sv 320 km downstream from the sill (Dickson and Brown 1994). At 480 km downstream from the sill, Dickson et al. (1990) and Dickson and Brown (1994) reported 10.7 Sv of deep transport. Dickson and Brown (1994), in reference to McCartney (1992), provided compelling arguments, suggesting that the difference between the observed transport (10.7 Sv) and the overflow transport, in concert with the Charlie Gibbs Fracture Zone transport (5.1 Sv + 2.4 Sv = 7.5 Sv), is largely caused by entrainment of recirculating cold, relatively fresh water from the Labrador Sea. Still farther downstream, off the tip of Greenland, Clarke (1984) estimated

Fig. 1. Topographic map of the North Atlantic Ocean. Source: NOAA 2012. The Mid-Atlantic Ridge running along the bottom of the North Atlantic is associated with plate tectonics and seafloor spreading. Near 52°N, the Charlie Gibbs Fracture Zone traverses the Mid-Atlantic Ridge, breaking it into northern and southern components. The northern extension that connects to Iceland is known as Reykjanes Ridge.
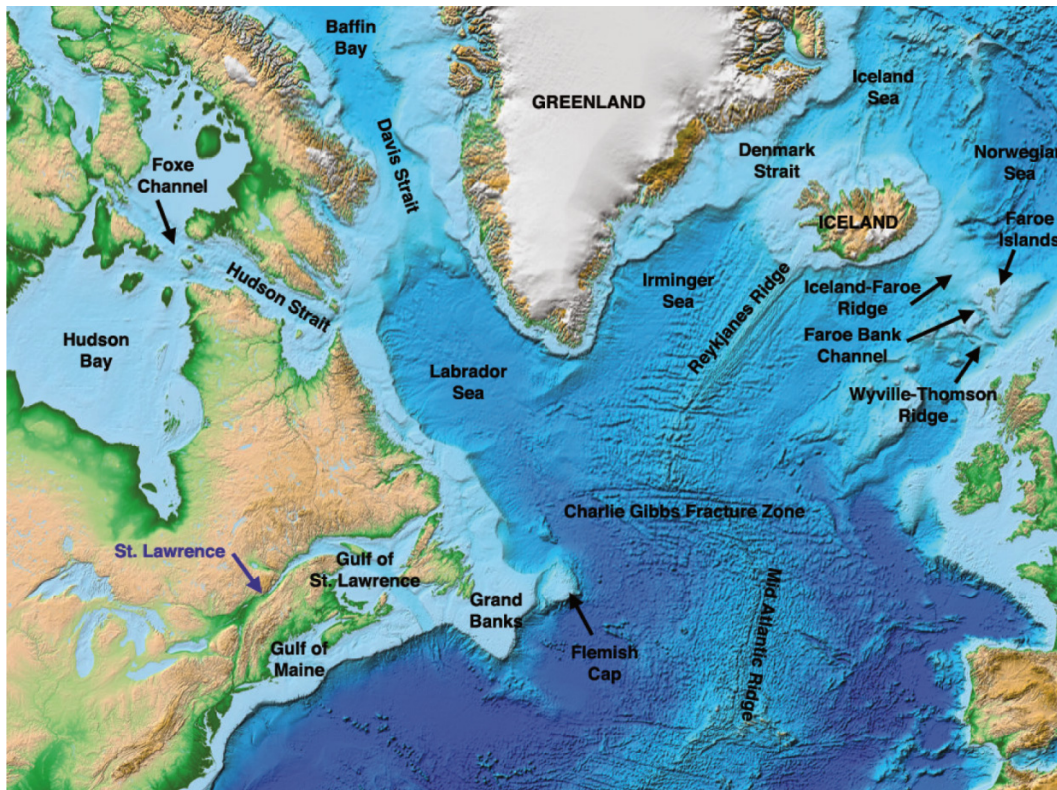
13.3 Sv of deep transport, an increase from that upstream by means of additional recirculating components and water mass entrainment. The deep western boundary undercurrent is thought to be 200–300 km wide and transport 13–14 Sv of newly formed North Atlantic Deep Water (NADW) (Warren 1981; McCartney and Talley 1984; Schmitz and McCartney 1993; Schmitz 1995) southward. These waters eventually encounter northward flowing Antarctic Bottom Water (AABW). Despite the high salinity of the NADW at such great pressures (further enhanced by mixing with Mediterranean water at mid-latitudes), the colder AABW has higher density and passes below the NADW.

Deep water also forms in the Labrador Sea; the depth at which it occurs in any given year is highly variable. Aside from two notable exceptions, convection depths in 1995–2005 were generally between 1000 and 1400 m. In 1995, convection extended downward to 2300 m, whereas in 2004 it was confined to 700 m (Avsic et al. 2006). Between 1987 and 1995, Yashayaev and Loder (2008) also noted a sequence of deep (~2300 m) convection years. Labrador Sea Water overrides NADW formed in the Nordic seas and is sometimes known as Upper North Atlantic Deep Water.

The surface circulation of the North Atlantic is characterized by the warm, saline northward-flowing Gulf Stream which, as it traverses the North Atlantic Ocean, becomes the North Atlantic Current. To the east of Reykjanes Ridge, the North Atlantic Current bifurcates into the northwestward-flowing Irminger Current as well as the North Atlantic Drift

that continues onward towards the northeast (Fig. 2). Cold, fresh Arctic waters are exported to the North Atlantic via the East Greenland Current. These are further fed by warmer, more saline Atlantic waters from the Irminger Current. Once it passes the southern tip of Greenland, the East Greenland Current becomes known as the West Greenland Current. A branch of the West Greenland Current flows northward into, around, and subsequently out of Baffin Bay (as the Baffin Current). Here, it meets with the southward-flowing Labrador Current (Fig. 2). The other branch of the West Greenland Current follows the topographic contours to the south of Davis Strait, eventually joining the Labrador Current. Cold, fresh, surface water flowing out of Hudson Strait also feeds the current.
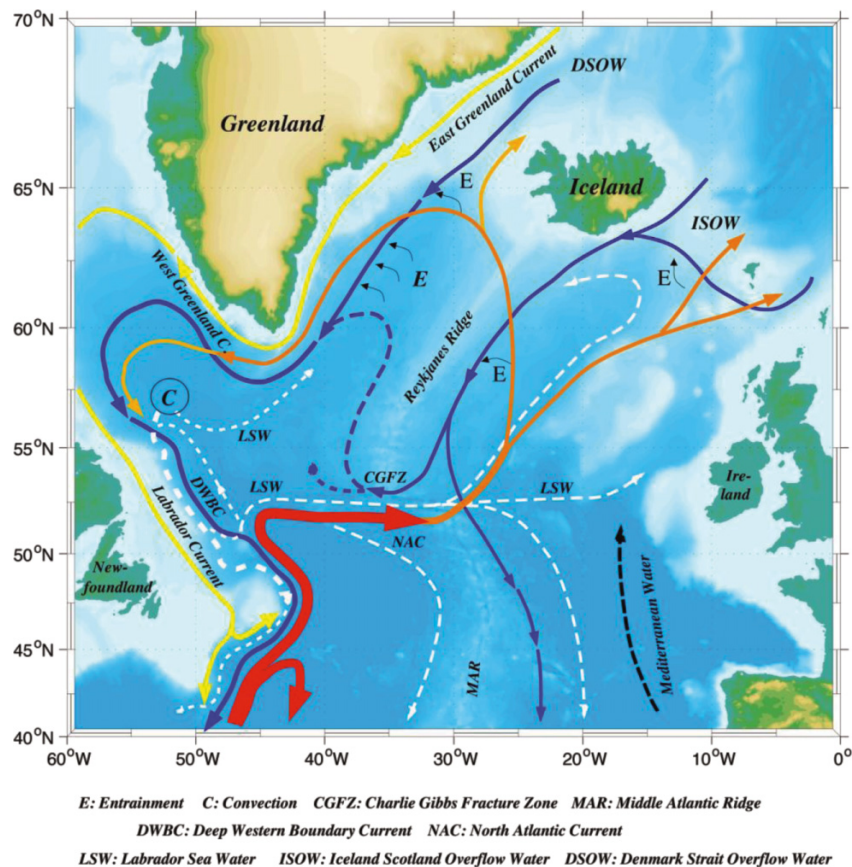
3.3. The Arctic Ocean

3.3.1. Geography

The Arctic Ocean is a semi-enclosed basin comprising 11.5 million km². It is bordered by six nations (Canada, US, Denmark (via Greenland), Iceland, Norway, Russia). The Lomonosov Ridge, rising to a minimum depth of ~950 m, separates the Arctic Ocean into the Amerasian and Eurasian basins. The Alpha–Mendeleev Ridge breaks the Amerasian Basin into the Canada and Makarov basins, while the Nansen–Gakkel Ridge partitions the Eurasian Basin into the Nansen and Fram basins (Fig. 3). The Arctic Ocean is connected to the Pacific Ocean by the shallow (~50 m deep), 85 km wide Bering

Fig. 2. The ocean circulation in the northern North Atlantic. Red, orange, and yellow lines show surface currents. Blue curves portray deep currents, while the dashed white line tracks water associated with the formation of the Labrador Sea Water mass. Source: www.ifm-geomar.de/index.php?id=19&L=1 (accessed 10–5–2011).

E: Entrainment   C: Convection   CGFZ: Charlie Gibbs Fracture Zone   MAR: Middle Atlantic Ridge
DWBC: Deep Western Boundary Current   NAC: North Atlantic Current
LSW: Labrador Sea Water   ISOW: Iceland Scotland Overflow Water   DSOW: Denmark Strait Overflow Water

Strait. There are both deep (Fram Strait) and shallow (Barents Sea) connections between the Arctic and North Atlantic Oceans. The Arctic Ocean is also connected to the North Atlantic Ocean via narrow channels through the Canadian Arctic Archipelago. The most significant of these are Lancaster Sound, north of Baffin Island, Cardigan Straight to the south of Ellesmere Island, and Nares Strait between Ellesmere Island and Greenland.

Two-thirds of Canada's coastline borders the Arctic Ocean, with the majority of this being associated with the islands of the Canadian Arctic Archipelago. Among these, Baffin Island (507 451 km^2), Victoria Island (217 291 km^2), and Ellesmere Island (196 236 km^2) are the fifth, eighth, and tenth largest islands in the world, respectively. Fresh water from land entering the Arctic Ocean is dominated by seven drainage systems. The Mackenzie River dominates runoff into the Arctic from the North American continent with a discharge rate of 281 km^3 yr^{-1} (ACIA 2005). Eurasian runoff is largely attributable to six river systems (Kolyma, Lena, Yenisey, Ob, Pechora, Severnaya Dvina); the Yenisey has more than twice the annual discharge of the Mackenzie River (580 km^3 yr^{-1}; ACIA 2005). The Kolyma, Lena, Ob, Pechora, and Severnaya Dvina discharge 103, 528, 402, 108, and 105 km^3 yr^{-1}, respectively (ACIA 2005).

The annual cycle of Arctic sea-ice extent is characterized by an end-of-summer minimum in September and an end-of-winter maximum in March. On 19 September 2010, the minimum sea-ice extent was the third lowest recorded since the satellite era (which began in 1979) (Fig. 4), dropping to 4.60 million km^2, or 2.11 million km^2 below the 1979–2000 median. On 7 March 2011, the maximum Arctic sea-ice extent matched the lowest recorded level since the satellite era (Fig. 4), reaching 14.64 million km^2, which was 1.2 million km^2 below the 1979–2000 median.

3.3.2. Circulation and water masses

A cold, fresh, surface layer overlying a warm, saline layer of Atlantic origin characterizes the water mass structure of the Arctic Ocean. The transition between these layers is marked by a distinct halocline and temperature inversion (the halocline occurs at the depth at which the rate of change in salinity with increasing depth is greatest). Arctic bottom waters are both cold and saline, with the boundary between the overlying Atlantic waters involving a weak thermocline (the thermocline occurs at the depth at which the rate of decline in temperature with increasing depth is greatest). The salinity characteristics of the upper layer vary across Arctic waters, depending on whether or not surface waters from the Pacific are present (McLaughlin et al. 2002). Driven by differences in temperature and salinity between the North Pacific and North Atlantic, which affect the density, volume, and height (steric height) of the oceans, ~0.8 Sv (1 Sv = 10^6 m^3 s^{-1}) flows into the Arctic Ocean via Bering Strait. This brings about 0.08 Sv of fresh

Fig. 3. Topographic map of the Arctic Ocean (International Bathymetric Chart of the Arctic Ocean). Source: IBCAO 2011 (Jakobsson et al. 2008).
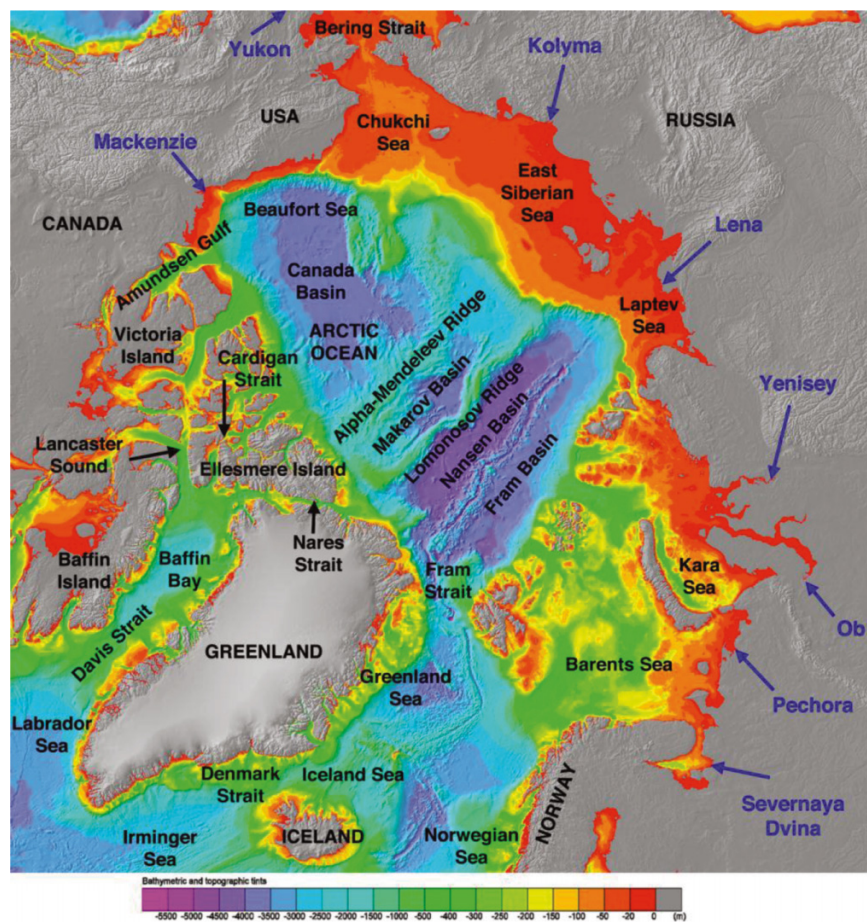

Fig. 4. Arctic sea ice extent at its 2010 minimum on 19 September and at its 2011 maximum on 7 March. The orange line shows the median sea ice extent over the period 1979–2000. Source: NSIDC 2012.
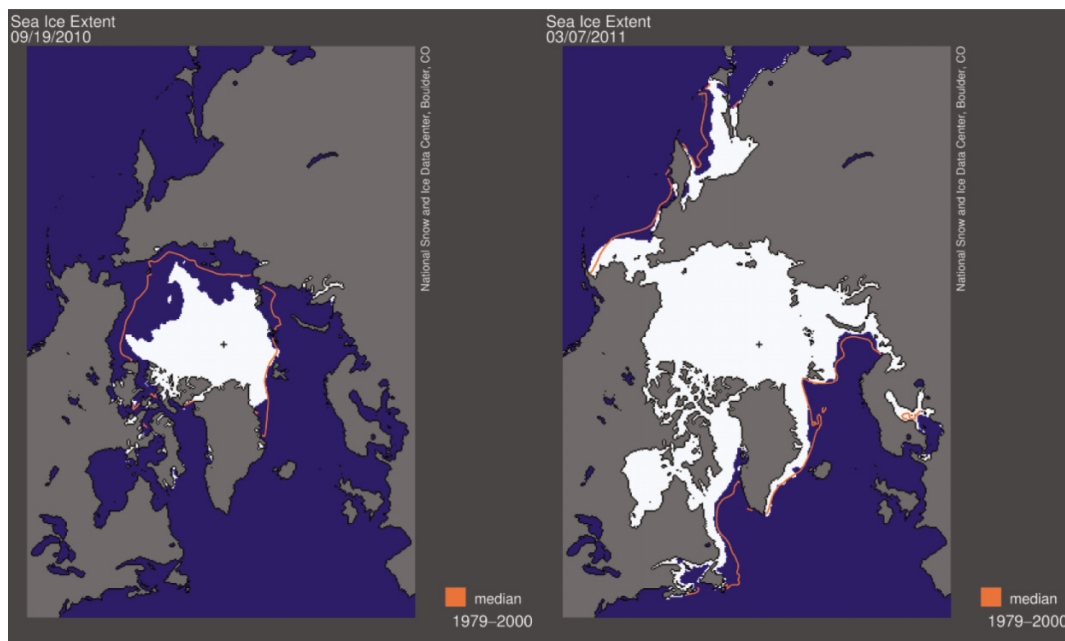
Fig. 5. Schematic diagram of the major surface (open blue arrows) and deep (red arrows) currents of the Arctic Ocean. Left: the anticyclonic regime. Right: the cyclonic regime. Source: Proshutinsky et al. (2005).
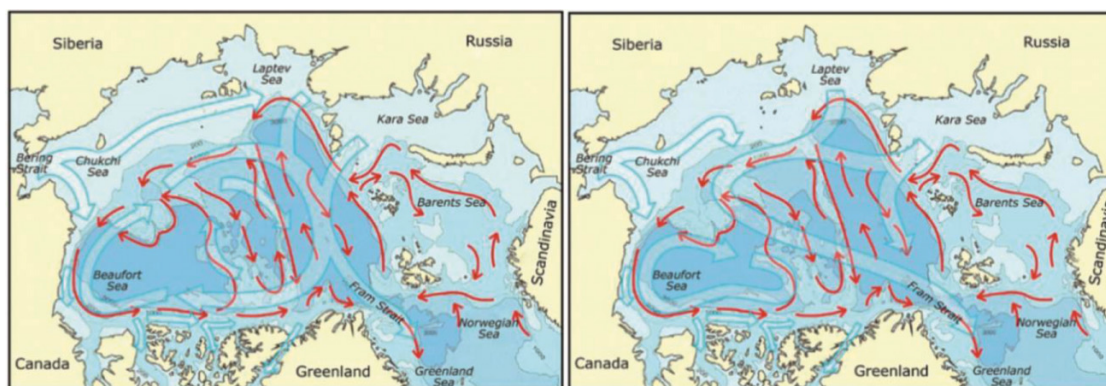
water from the Pacific into the Arctic. Freshwater export to the Atlantic occurs through the Canadian Arctic Archipelago and Fram Strait (Melling et al. 2008).

Warm, subsurface Atlantic waters enter the Arctic via two distinct branches: Fram Strait and Barents Sea (Fig. 3; Jones 2001). These flow along continental slopes around the Arctic and its deep ocean basins in a counter-clockwise fashion (Rudels et al. 1994). The surface circulation of the Arctic Ocean is generally clockwise (Fig. 5), albeit highly variable. The Beaufort Gyre in the Beaufort Sea and the Transpolar Drift from the Siberian Shelves to the Atlantic Ocean are the dominant features of this circulation. The Beaufort Gyre is particularly important to the climate system because it contains a vast reservoir of fresh water (Proshutinsky et al. 2002, 2009). Proshutinsky and Johnson (1997) demonstrated that the wind-driven surface circulation of the Arctic Ocean is characterized by two differing regimes. Fresh water is generally exported from the Beaufort Gyre to the North Atlantic in the cyclonic (counter-clockwise) regime (Proshutinsky et al. 2005; left panel in Fig. 5) and accumulates in the Beaufort Gyre in the anticyclonic (clockwise) regime (Proshutinsky et al. 2005; right panel in Fig. 5).

In 2009, winds over the Arctic drove annual-mean cyclonic surface flow that greatly weakened the Beaufort Gyre and the transpolar drift (Proshutinsky et al. 2010). In addition, the seasonal cycle of the wind-driven circulation reversed relative to climatology, becoming anticyclonic in summer rather than cyclonic, and cyclonic instead of anticyclonic in winter.

3.4. The northeast Pacific Ocean

3.4.1. Geography
The northeast Pacific Ocean off British Columbia (BC) is bounded to the east by a rugged coastline with steep and complex topography. Vancouver Island and the mainland's Coast Mountain Range, the somewhat less rugged but still complex topography in Haida Gwaii, and numerous smaller islands closer to the mainland are the predominant topographical features (Fig. 6). The mainland coast is dissected by a complex network of inlets, straits, passes, sounds, and narrows. From the border of Washington State to the Alaska panhandle, the coastline, inclusive of islands, is almost 27 300 km long (Thomson 1981).

The continental shelf bordering the coast and islands is relatively broad in Queen Charlotte Sound and Dixon Entrance but is especially narrow on the west coast of Haida Gwaii. On the west coast of Vancouver Island, the shelf is relatively narrow in the north but relatively broad near the exit of the Strait of Juan de Fuca. Three major areas of deep bathymetry on the continental shelf are located at Dixon Entrance, Queen Charlotte Sound, and the Strait of Juan de Fuca.

The slope connecting the continental shelf with deeper offshore waters is relatively steep and narrow compared with the slope fringing the continental shelves in the Arctic and northwest Atlantic oceans. There are about 30 steep-walled canyons cutting across the slope between Cape Flattery to the south and Cape St. James to the north (Thomson 1981).

Moving offshore, depths gradually increase, with the exception of numerous inactive subfloor volcanoes called seamounts and some broad underwater ridges, which are characterized by peaks and valleys. Bowie, Union, and Cobb seamounts are among the most prominent undersea seamounts off BC and are part of a cluster of approximately 100 that extend from the Gulf of Alaska to the Oregon coast (Thomson 1981). Parts of these seamounts rise to within the sunlit portion of the upper ocean waters and are considered "hotspots" for sea life.

3.4.2. Circulation and water masses
The large-scale current systems of the North Pacific are associated with oceanic gyres in the Bering Sea, the western Subarctic, the Gulf of Alaska, and subtropical Central Pacific areas (Fig. 7). The North Pacific Current is a broad, slow-moving eastward extension of the Kuroshio Current that lies to the south of the Subarctic Boundary. The Subarctic Boundary is an oceanic front that separates the relatively warm, high-salinity, and low-productivity waters of the subtropics from the cooler, fresher, nutrient-rich, and more productive waters of the Subarctic North Pacific. The West Wind Drift is part of the Subarctic Current system, which bifurcates into the northeastward-flowing Alaska Current and the southeastward-flowing California Current as it nears the Gulf of Alaska. The area of divergence between the Alaska and California currents typically has variable currents that include many eddies and meanders that range in size from tens to hundreds of kilometres.

Fig. 6. Topographic and bathymetric map of the northeast Pacific Ocean. Source: NOAA 2012.

Fig. 7. Prevailing surface currents in the North Pacific Ocean. Double arrows are intense boundary currents with speeds typically 1–2 m s^{-1}; over most of the region, speeds are less than 0.25 m s^{-1}. Broken arrows correspond to the winter Davidson Current off the Oregon–Washington coast. The asterisk off the coast of BC is the location of Ocean Station PAPA, a weather ship, from 1957 to 1980. Source: Thomson (1981).
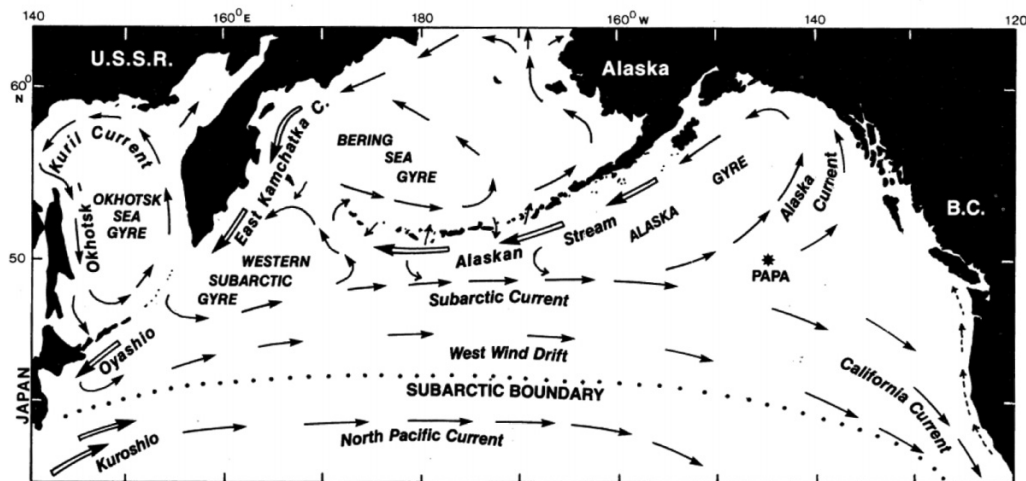
3.5. Biological oceanography: some basics

Marine ecosystems, like terrestrial ones, are composed of a variety of living organisms and physical attributes that interact through a sequence of processes involving the production and transfer of energy (Kaiser et al. 2005; Nybakken and Bertness 2005; Castro and Huber 2007). Energy from the sun is captured by autotrophs (e.g., plants, bacteria, and algae) and is stored in the chemical bonds of organic compounds. Heterotrophic organisms obtain energy by eating autotrophs (herbivores), eating other organisms that have eaten autotrophs (carnivores), or absorbing dissolved organic matter from the environment. This arrangement of autotrophs and succeeding levels of heterotrophs defines a trophic structure (often illustrated as a food web), a characteristic feature of all ecosystems. The final component of the trophic structure consists of the detritivores and decomposers. Detritivores are multicellular organisms that consume fragments of dead organisms. Decomposers, typically bacteria and fungi, break down the complex organic compounds of dead organisms. Simple molecules are thus released in the form of dissolved organic

Fig. 8. Three marine ecosystem zones (littoral intertidal, pelagic, benthic) in relation to depth.
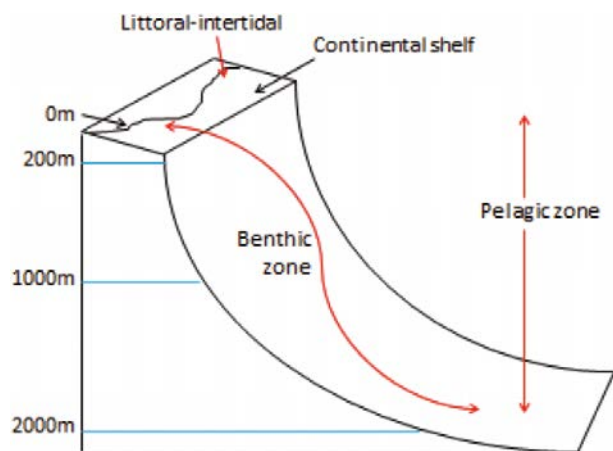
carbon which is again exploited within the food web. This is a unique feature of aquatic food webs. In addition to an energy source (generally, but not exclusively, sunlight), marine ecosystems require nutrients, of which nitrogen, phosphorus, iron, and silicate are the most important.

The marine realm may be viewed as a hierarchical arrangement of such ecosystems (Fig. 8). Five major categories of ecosystems are recognized in marine waters: pelagic, benthic, littoral intertidal, estuarine brackish, and deep sea. The pelagic realm encompasses the water column away from the bottom and the shore. The upper pelagic (or epipelagic) ecosystem, comprising the first 200 m of the water column, is the warmest and receives the most sunlight of the pelagic realm. It thus includes the photic zone where light is sufficient to permit photosynthesis. The benthic realm includes organisms associated with the seabed that are collectively known as benthos. The shallowest part of the benthic realm is the littoral-intertidal zone, the narrow fringe along the shoreline found between the highest high tide and the lowest low tide. Although accounting for the smallest area of the world's oceans, they are the best known of marine ecosystems because they are so readily accessible for study. The intertidal zone is unique among marine ecosystems because it is regularly exposed to air and experiences the greatest variation in environmental factors. The estuarine-brackish realm comprises an ecosystem created when fresh waters flowing from rivers first meet and mix with salt water from the sea. These systems are among the most productive environments on the planet, ranking alongside tropical rain forests and coral reefs. The deep sea ranges from the edge of the continental shelf at about 200 m depth down to the abyssal plain 5 km below the surface, with some deep trenches continuing down to a depth of 10 000 m. Of the 70% of the planet's surface covered with water, about 85% of the area constitutes the deep sea. Although inhospitable to most forms of life because of massive pressure, near-freezing waters, and a total lack of sunlight, the deep sea is believed to harbour a huge yet largely unexplored biodiversity (Webb et al. 2010).

Each of these realms (pelagic, benthic, littoral intertidal, estuarine brackish, deep sea) can be subdivided into more discrete functional units characterized by specific physical conditions, including light and available nutrients which vary in concert with, among other attributes, depth, water clarity, salinity, and temperature. In addition, there are geographical differences in the communities of living organisms composing these functional units, reflecting in part the evolutionary history of different parts of the marine realm. The resulting geographical patterns of this biodiversity and their associated food webs contribute to the identification of marine ecoregions.

3.6. Canadian marine ecoregions

Marine ecoregions are biogeographic classifications of patterns of biodiversity. A major objective of biogeography is to identify and characterize geographic groupings of species and the biogeochemical conditions that make them different (Longhurst 1998). Thus, marine ecoregions are defined at the scale of the continental shelf according to a combination of geological, physical oceanographic, and biological properties. Ecoregions may be grouped within larger marine areas known as ecoprovinces. Here, we adopt the scheme proposed by Powles et al. (2004) and adopted by DFO (2009) in the national framework for Canada's proposed network of marine protected areas. Canadian marine waters encompass three marine ecoprovinces and twelve ecoregions: three in the Cold Temperate northwest Atlantic; four in the Cold Temperate northern Pacific; and five in the vast Arctic realm. Coastal and shelf waters, combining benthic and shelf epipelagic biotas, represent the areas in which most marine biodiversity is found.

At times, ecoregion boundaries coincide with major biogeographical discontinuities. For example, in the Pacific Ocean, the Southern Shelf ecoregion (ecoregion 2; Fig. 9) is bordered in the north by the Brooks Peninsula which divides the continental shelf at this point. The Brooks Peninsula represents the northern distribution of many marine species. To the north of the Brooks Peninsula is the Northern Shelf ecoregion (ecoregion 4; Fig. 9). A distinctive feature of this ecoregion is the shallow-water area located between Haida Gwaii and the mainland. Many species and populations in this ecoregion do not extend to the south of the Brooks Peninsula. For example, all major bird colonies in British Columbia occur north of the Brooks Peninsula.

Some ecoregions are clearly defined by depth and their position on or off the continental shelf. In the Arctic Ocean, the Arctic Basin ecoregion (ecoregion 5; Fig. 9) is located off the continental shelf and is characterized by depths greater than 1000 m. Primary production is considered low in this ecoregion due to permanent ice cover. In contrast, the Western Arctic (ecoregion 6; Fig. 9) is a relatively shallow region, encompassing the Beaufort Sea to the west, Amundsen Gulf, Queen Maude Gulf to the southeast of Victoria Island, and Viscount Melville Sound to the north of Victoria Island (Fig. 3). The ecoregion is characterized by pack ice in the north and seasonal ice in the south, although these patterns are rapidly changing as the Arctic Ocean warms. The region is also relatively shallow (less than 200 m) and has two particularly shallow areas, one being the Queen Maude Gulf and the other located between Viscount Melville and Lancaster Sound to the east. This latter shallow-water boundary is an important biological boundary between eastern and western populations for many species.

Fig. 9. Canadian marine ecoregions. Source: DFO (2009).
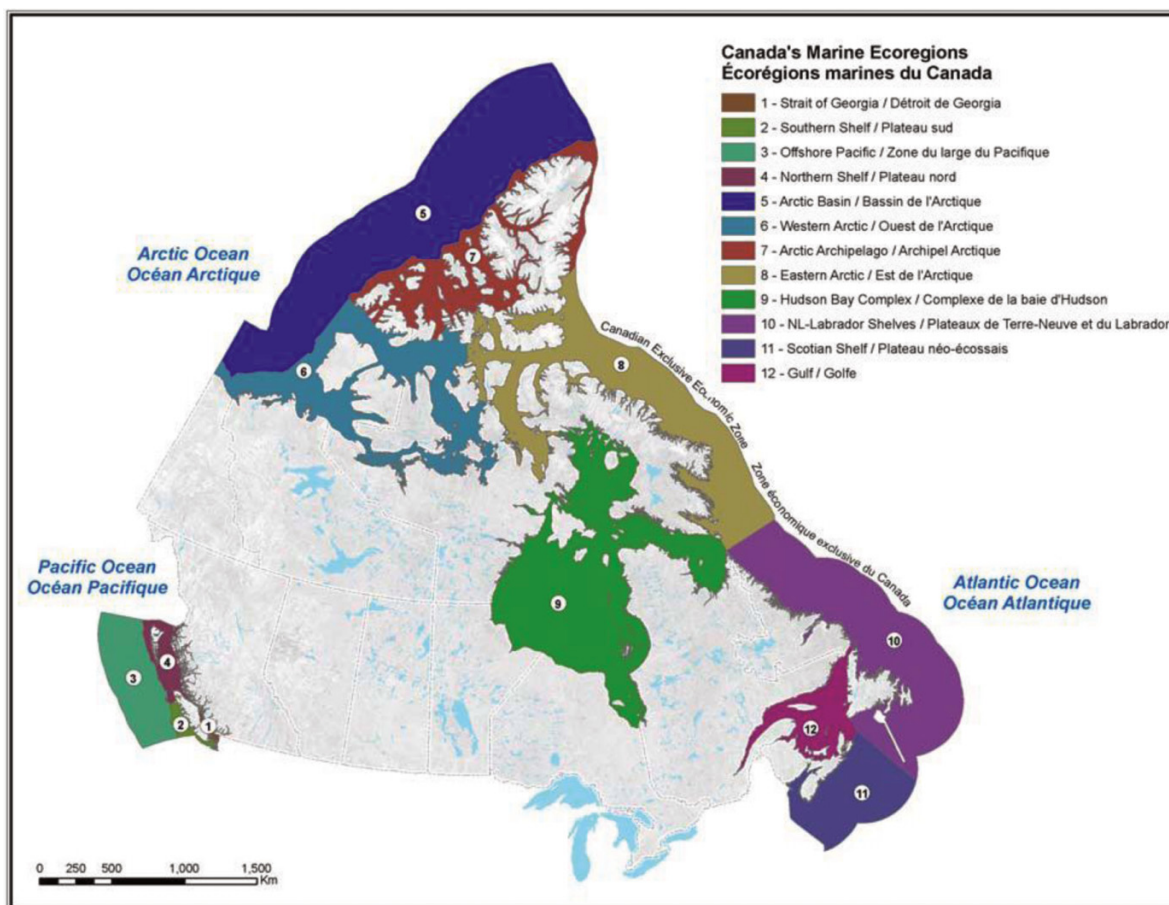
Enclosed marine areas also represent unique ecoregions because of the dominance of tidal mixing and freshwater outflow from adjacent rivers. The Hudson Bay Complex ecoregion (ecoregion 9; Fig. 9) is characterized by its degree of closure and relatively shallow depth. Tides are an important oceanographic feature here, as is the large input of fresh water from Québec rivers. Ice cover is seasonal and primary production is generally low, although higher productivity is evident in Hudson Strait and Foxe Basin, as result of strong tidal mixing. In the Atlantic ecoprovince, the primary feature of the Gulf of St. Lawrence ecoregion (ecoregion 12; Fig. 9) is water flow, which is essentially continuous from the Strait of Belle Isle in the north, through the Gulf, and onto the eastern Scotian Shelf. Freshwater influence is important, with the St. Lawrence River flowing eastward across the southern Gulf and onto the Scotian Shelf. The southern Gulf has warmer water temperatures due to its shallowness and has greater stratification than adjacent waters. The St. Lawrence estuary has colder water, but strong mixing does result in high primary productivity at the mouth of the river.

These 12 marine ecoregions define Canada's marine landscape. They demonstrate spatial patterns in ecosystem processes as well as geographical differences in the communities of living organisms comprising Canadian marine ecosystems. Mapping biogeographic patterns in this way is

an essential step in understanding the richness, functionality, and distribution of Canadian marine biodiversity and planning an integrated management approach that will take into consideration the regional impacts arising from climate change and differing sectors of human activity, including fisheries and aquaculture.

However, important as these ecoregions are for biogeographic purposes, they belie an absence of data at finer spatial scales on the biological use of marine habitats for most Canadian species. Although efforts have been made to document these habitats (e.g., Stewart et al. 2001) and increase the spatial resolution of benthic habitat mapping (e.g., using multibeam acoustic mapping of the seafloor; Smith et al. 2009), temporal data on the physical and biological characteristics of Canada's marine habitats are lacking. For example, according to one of the submissions received by the Expert Panel, Canada lacks time series data of the areal extent of marine macrophytes, a deficiency that might be explained by the assertion (from the same submission) that there is only one individual in DFO responsible for undertaking near-shore habitat mapping on bay-wide scales. It is to be hoped that Stephenson and Hartwig's (2010) recent mapping efforts to identify areas of high biological importance in the Arctic will be repeated elsewhere.

Table 1. Marine species richness for select taxonomic groups and habitats in Canada.

Taxonomic group or habitat	Estimated number of species		Source
	Canada	Global	
Microbes (Arctic only)	9500–54000	Unknown	Archambault et al. (2010)
Phytoplankton	1657	~5000 (25000)	Archambault et al. (2010)
Macroalgae	860–979	~9300	Archambault et al. (2010); Mike Guiry (www.algaebase.org)
Cold-water corals	104	700	Campbell and Simms (2009)
Sponges	265	5000–10000	Susanna Fuller (personal communication, 2011); Austin et al. (2010); Fuller (2011)
Zooplankton	900	Unknown	Archambault et al. (2010)
Benthic infauna	2127	Unknown	Archambault et al. (2010)
Fishes			
Cartilaginous	61	~1100	Nick Dulvy (personal communication, 2011)
Bony	831–971	14200	Archambault et al. (2010)
Seabirds	38–64	383–475	I. Jones and R. Cannings (personal communication, 2011); www.cornell.edu; Clementschecklist; Cheung et al. (2011a)
Mammals	52	125	Archambault et al. (2010)
Hydrothermal vents	66	592	Bachraty el al. (2009)

3.7. Species richness of Canada's oceans

Canada's geography has mixed effects on the biodiversity of marine species. On one hand, Canada is disadvantaged because it is a northern country. With few exceptions, species richness (the number of species in a given area) becomes poorer as latitude increases. This trend is particularly marked in the northern hemisphere because of the relatively young age of the Arctic, which has afforded comparatively little time for speciation and endemism (Dayton 1994). On the other hand, the areal extent of Canada's oceans is vast (~7.1 million km^2); more species are typically found in larger areas. The species richness of Canada's oceans is also likely to be enhanced by the heterogeneity of the country's marine realm, given the country's location at the junction of three oceans, each of which contributes its own fauna and flora to national marine biodiversity.

Species richness is often the most practical means of considering biodiversity, given that it describes well the variety of life that people encounter. However, it is, of course, only one facet of biodiversity (section 2). One important consideration is that species richness is generally not as sensitive to direct human impacts as are other aspects of biodiversity, such as abundance and distribution. For example, even though Atlantic cod has been severely depleted (Hutchings and Rangeley 2011) and its core distribution has been much reduced (Hutchings 1996), it still contributes to the richness of the Atlantic fish community, as do species that have never declined to historically low levels of abundance.

The question of how many species occur in any relatively large area cannot be answered with great certainty. No comprehensive inventory of marine species exists for any part of the world. Even in regions such as Europe, which has a relatively extensive marine species list, it is estimated that up to 33% of species remain to be described (Costello and Wilson 2011). This means that for every 10 European marine species known, at least three more are not. Globally, the proportion of described marine species is likely much lower. Mora et al. (2011), for example, estimated that 91% of their estimated 2.2 million marine species have yet to be described. The overall proportion of species remaining to be discovered is undoubtedly higher for places such as Canada, where the national marine inventory is much less complete. Nevertheless, species richness is relatively well established for some marine taxa in Canadian waters. The aim here is thus to give an appreciation, rather than a complete accounting, of the natural biological wealth of Canadian coastal waters. This is achieved by (*i*) presenting information on the better-known groups to illustrate the patterns of richness of Canada's three oceans, (*ii*) comparing them to patterns found in Canadian terrestrial fauna and flora, and (*iii*) estimating the numbers of species yet to be discovered.

3.8. How many species are in Canada's oceans?

The recent global Census of Marine Life programme prompted an attempt at cataloguing the biodiversity of the world's oceans, including Canada's (Archambault et al. 2010). The total number of species in Canadian waters enumerated by 2010 reached a minimum of 15 988 (Table 1). This increases to almost 16 500 species by adding (*i*) an estimated 38 (known to breed in Canada; Cheung et al. 2011a) to 64 species of seabirds (Dr. Ian Jones, Memorial University of Newfoundland, and Dr. Richard Cannings, Okanagan Valley, BC, personal communications, 2011), which are considered here because of their near-exclusive reliance on marine foraging resources; (*ii*) 265 species of sponges (Austin et al. 2010; Fuller 2011; Dr. Susanna Fuller, Ecology Action Centre, Halifax, NS, personal communication, 2011); (*iii*) 104 species of cold-water corals (Campbell and Simms 2009); and (*iv*) 66 species of invertebrates and fishes recorded to date in exclusive association with deep-sea hydrothermal vents off BC (Bachraty et al. 2009).

This overview is, by necessity, incomplete. It focuses on a few taxonomic groups for which information is both sufficient and accessible. Thus, marine fishes, birds, and mammals are relatively well enumerated, but significant gaps remain for other taxa and habitats that are known, or suspected to be,

species-rich. Marine microbes, for example, have to date only been enumerated in the Canadian Arctic, where only a fraction of the expected full species diversity has been described. Although an old adage holds that at least half of the species in the world are parasites, there are currently no parasites or other symbionts on the Canadian marine list. The list of invertebrates associated with the sea floor includes only those organisms that live buried in sediment (i.e., infaunal invertebrates), but not those attached to the bottom. Moreover, the spatial extent of sampling for infaunal invertebrates has been minuscule, amounting to only 248 m^2 of the sea floor (much less than one millionth of 1% of Canada's continental shelf) surveyed to date (Archambault et al. 2010). Many species of animals and algae associated with rocky intertidal habitats are missing from the tally, as are those that depend on cold-water coral and sponge reefs, because information pertaining to these habitats is either sparse or nonexistent.

Even for groups for which there is a Canadian total, their numbers may, to various degrees, be underestimated. For example, Shackell and Frank (2003) estimated that, since the mid-1970s, one to six new species are added annually to the list of fishes caught in DFO bottom-trawl surveys on the Scotian Shelf. The degree of error for macroalgae is even more extreme. Recently, genetic studies have uncovered 150–200 new species of macroalgae in Canada with most hailing from the Pacific coast (Dr. Gary Saunders, University of New Brunswick, NB, personal communication, 2011). These recent discoveries are estimated to boost the current species richness of Canadian macroalgae by 15%–23% (Table 1).

3.9. Patterns of species diversity

The current assessment of Canadian marine biodiversity, however incomplete, still allows a glimpse of the relative distribution of species on the country's three coasts. Some of the patterns can almost certainly be attributed to regionally uneven research effort. For example, it might seem surprising that the overall number of species in the Arctic is similar to that of the other two coasts (Fig. 10). However, this is in large part due to the abundance of Arctic phytoplankton species (especially diatoms in the subkingdom Chromista; Fig. 10A) that have been the subject of compilations for the polar region, but not for the other two coasts.

Other patterns may reflect true differences in species richness. For example, crustaceans, notably harpacticoid copepods and infaunal amphipods, appear to be more numerous in Arctic waters than elsewhere, despite lower sampling efforts in the north. The Pacific coast is likely to be a true national hotspot of macroalgae diversity (Fig. 10B), with at least 650 species enumerated to date (not including recent discoveries; see above). In addition, the enumeration of species of vertebrates is considered to be relatively complete. Thus, fish species richness peaks on the Atlantic coast (Fig. 10C), a pattern which might be influenced in part by the relatively limited extent of Canada's Pacific coastline. The pattern of seabird diversity mirrors that for marine mammals, with both being low in the Arctic and similarly high on the east and west coasts. It is also notable that Canadian oceans are home to more than 40% of the world's marine mammal species.

3.10. Canadian species diversity in a terrestrial context

How does Canadian marine species diversity compare with its terrestrial counterpart? At nearly 10 million km^2, the surface area of Canada is almost 1.5 times greater than that of Canadian marine waters. To date, scientists have enumerated ~71 000 Canadian species (Mosquin et al. 1995), ranging from freshwater microorganisms to the giant, western red cedars (*Thuja plicata*) of Pacific coastal rainforests. This represents approximately 4.4 times more species than are currently identified within Canada's oceans. This unevenness varies across taxonomic groups. For example, terrestrial birds outnumber seabirds by approximately 8 to 1 (land versus sea: 398 versus ~50), yet there are only about 3 times as many Canadian terrestrial mammal species as there are marine species (148 versus 52). Among fishes, the pattern is reversed, with some four marine species for every freshwater species (~900 versus ~204).

The overall higher richness of Canadian terrestrial and freshwater taxa may be real. After all, the most species-rich group — insects — is only found on land. In Canada, as elsewhere, insects dominate species tallies, typically contributing more than 40% of species. However, at least some of the apparent inequality might be attributed to poorer sampling effort and capacity to describe marine biodiversity. It is widely acknowledged, for example, that diversity in the midwater and deeper parts of the oceans is generally poorly known (Webb et al. 2010) and likely to be very high. It is also telling that only 16% of Canadian researchers with taxonomic expertise surveyed by the Council of Canadian Academies' Expert Panel on Biodiversity Science (2010) reported marine habitats as the main habitat for their primary taxa of expertise. In contrast, 61% reported focusing on aspects of terrestrial biodiversity and 23% on freshwater taxa, and more than a quarter of taxonomic experts in Canada work on insects. The current inequality in species diversity between land and sea is reversed, however, if one considers higher taxonomic levels. For example, at the phylum level, Canadian oceans are far more diverse than land; two thirds of the 63 major phyla found in Canada are predominantly marine, reflecting the much longer evolutionary history of marine versus terrestrial organisms.

3.11. How many more species?

Canada's three oceans are very diverse. However, our understanding of this diversity is uneven. More is known about vertebrates than invertebrates, about large than small organisms, and about swimming than burrowing creatures. It is very likely that the most serious gaps in our knowledge of species diversity in Canadian waters pertain to the most species-rich groups (e.g., the smaller infaunal invertebrates). A more complete understanding of the species richness of Canada's oceans will require significant shifts in research focus and societal interest.

The number of species in Canada's oceans that have yet to be discovered can be estimated by various means. One is to consult taxonomic experts who can provide educated estimates of the number of species that remain to be recorded or described in their taxa of interest. This was done by Mosquin et al. (1995), who estimated that only 48% of marine species in Canada had, to that date, been named and classified.

Another method is to extrapolate from a taxonomic group, or a region, that has been relatively well surveyed. Marine

Fig. 10. Species richness of marine microbes, plants, and animals in the (A) Canadian Arctic, (B) Pacific, and (C) Atlantic. Data compiled from Archambault et al. (2010) with additions from Campbell and Simms (2009) (Cnidaria) and from personal communications (2011) with I. Jones and R. Cannings (seabirds, Aves). Pisces includes cartilaginous and bony fishes.
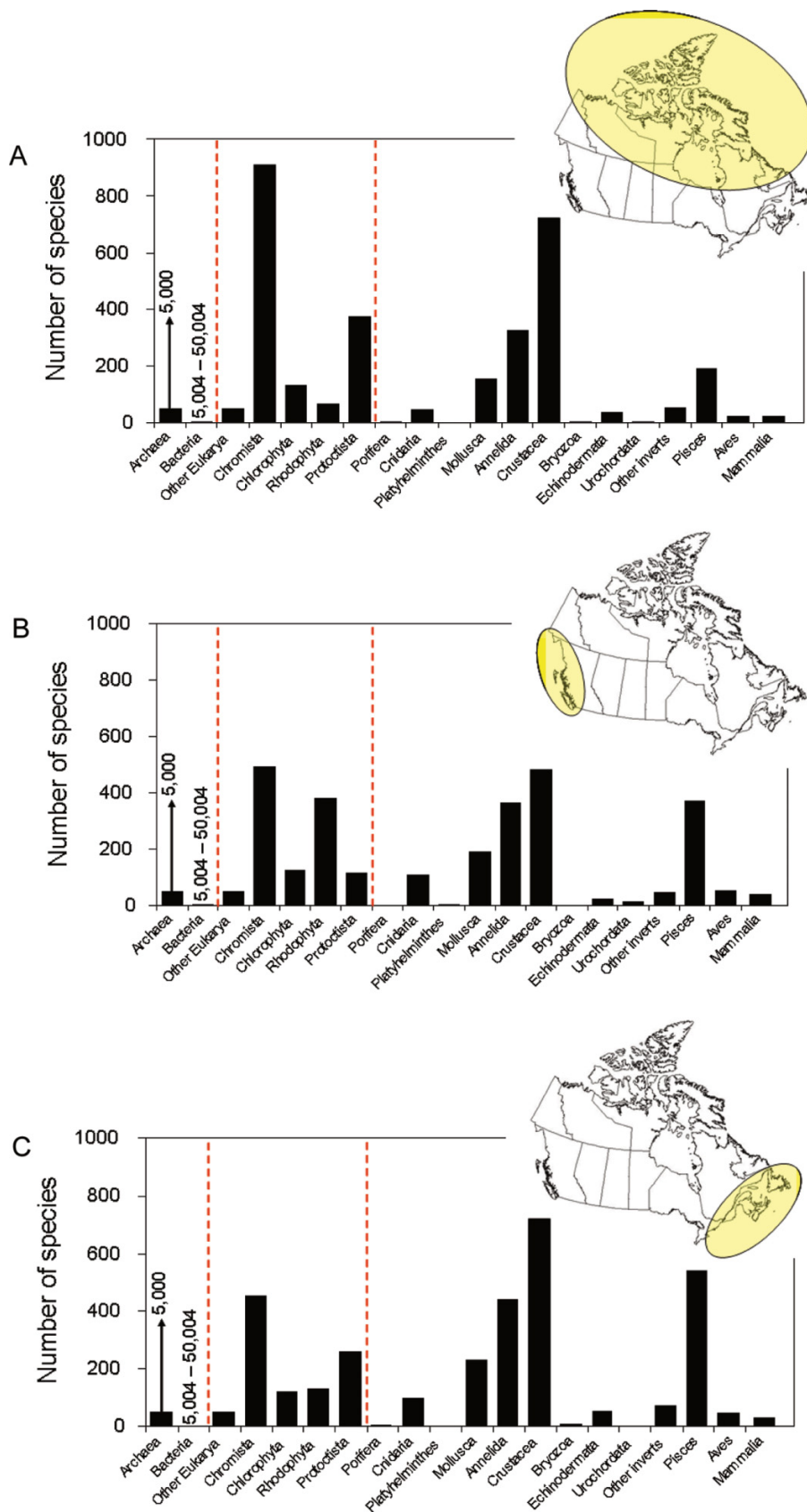
Fig. 11. Accumulation of marine invertebrate species around Haida Gwaii, BC, in relation to the number of literature- or collection-based records obtained between 1878 and 2000. Redrawn from Sloan and Bartier (2009).
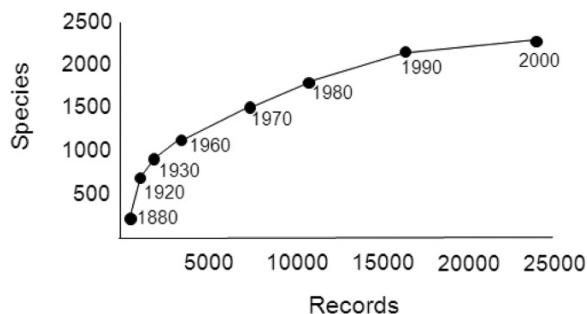
fishes within European seas are good examples. There are currently 27 929 recorded species of multicellular plants and animals in the European Register of Marine Species (Costello and Wilson 2011). This list includes a much more comprehensive coverage of habitats and area than its Canadian counterparts. Of this total, 1349 species are fishes. If fish species comprise the same proportion of species in Canada as they do in Europe, the ~900 Canadian marine fishes should be associated with ~18 600 plant and animal species in Canada's oceans (2.7 times as many as have been identified to date). Of course, this extrapolation will be biased if the ratio of fish to total richness varies geographically (which it probably does) or the reference inventory is incomplete (which it certainly is).

A third approach is to consider the rate at which new species are added to an inventory as sampling effort increases or time elapses. The premise of this method is that new species should initially be added quickly, given that there are many unrecorded species available. However, the rate of addition of new species should decrease as samples accumulate and time elapses, as scientists continue to search for elusive, unrecorded, or undescribed species. The relationship between the cumulative number of species in an area and effort or time should theoretically plateau to reveal the total number of species present. For example, when applied to the marine invertebrates of Haida Gwaii (Pacific Northern Shelf ecoregion), arguably the best inventoried marine subregion of the west coast (Sloan and Bartier 2009), the predicted number of species for this relatively small area exceeds 2250 (Fig. 11), nearly twice as many as that shown in Fig. 10B for the entire Canadian Pacific region (Archambault et al. 2010).

4. Physical and chemical indicators of climate change in Canada's oceans

4.1. Introduction

The primary purpose of this section is address the question in the panel's terms of reference that asked, "What are the past and current trends and associated uncertainties in physical and chemical indicators of climate change in Canada's three oceans?" The section begins with an examination of past and projected trends in several surface properties of the oceans (and, on occasion, comparative analyses of the terrestrial environment), including temperature, precipitation, salinity, and sea ice. This is followed by an examination of temporal trends in regional wind systems. Thereafter, the section focuses on

trends in indices that reflect ocean climate variability. To take one of these indices as an example, when the Pacific Decadal Oscillation is positive, the west Pacific cools and parts of the eastern Pacific warm. The section concludes with treatments of temporal patterns in coastal sea levels, water chemistry (including ocean acidification), and ocean stratification. Although the primary focus is on Canadian waters, some of the trends are presented at a global scale either for comparative purposes or because of the difficulty in extrapolating projections derived from global models to the much smaller spatial scale of Canada's marine environment. The potential consequences of these past, current, and projected trends in these physical and chemical indicators of climate change are examined in section 7.

4.2. Surface properties

### 4.2.1. Observed 20th century changes

Both globally and annually, averaged ocean surface temperatures have increased at a rate of 0.07 °C/decade over the past century, but by 0.17 °C/decade during the last thirty years (NOAA 2010). The spatial pattern of this annually averaged warming trend reveals several characteristic features (Fig. 12): (*i*) there have been greater rates of warming over land than over oceans (In the northern hemisphere, terrestrial warming rates have generally been greatest in winter for North America and Europe, and in spring for Asia; Fig. 13); (*ii*) there is more warming in the northern hemisphere than in the southern hemisphere; (*iii*) the warming rates are typically stronger in the middle of the continents or on their leeward coasts than on their windward coasts; (*iv*) the warming rates at high latitudes are greater than those at low latitudes; (*v*) there have been some localized regions of cooling.

The northwest Atlantic annual mean surface air temperature trend over the last century (left panel of Fig. 12) is also discernible from localized station temperature data in the Gulf of St. Lawrence and on the Newfoundland Shelf (Figs. 14, 15). As noted by Galbraith et al. (2010), the warming trend inferred from Charlottetown surface air temperature data is 0.78–0.90 °C per century, whereas it is 2.0 °C per century for equivalent data from Pointe-au-Père (Fig. 14). Although data from Station 27 on the Newfoundland Shelf show very little warming since 1950 (Fig. 15), the rate of warming since the mid-1980s is significant, both within the Gulf of St. Lawrence and at Station 27. Adequate data are not available to allow for the reliable calculation of precipitation trends over many parts of the world (Fig. 16). However, those features of the climate system that integrate precipitation falling over wide surface areas, and for which long-term records exist (such as sea surface salinity (SSS) and river discharge), provide valuable indicators of changes in the hydrological cycle and, in particular, its intensification. For example, Peterson et al. (2002) reported that discharge into the Arctic Ocean from the six largest Eurasian rivers (see Fig. 3) had increased by 7% from 1936 to 1999. In addition, Durack and Wijffels (2010) examined trends in global ocean surface salinity changes from 1950 to 2008 (Fig. 17). Their analysis revealed strong trends in SSS over much of the global ocean. The researchers noted that the pattern of these spatially coherent trends bore a striking resemblance to the climatological SSS field (Fig. 17B; cf. Fig. 17A). Those high SSS subtropical regions dominated by

Fig. 12. Observed trends in annual mean temperatures over the period 1901–2005 (left) in °C per century and 1979–2005 (right) in °C per decade. Red regions indicate warming trends; blue regions indicate a cooling trend; grey regions indicate insufficient data to determine a trend reliably. Source: Trenberth et al. (2007).
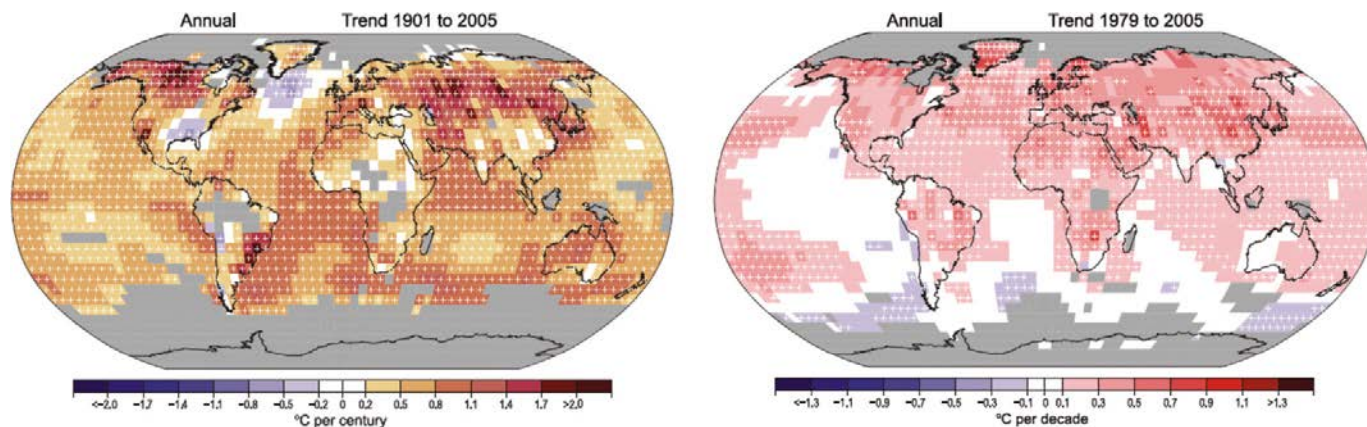
Fig. 13. Trends in seasonal mean temperatures over the period 1979–2005 in °C per decade. Red regions indicate warming trends; blue regions indicate a cooling trend; grey regions indicate insufficient data to determine a trend reliably. Results are shown for the seasons spring (March, April, May: MAM), summer (June, July, August: JJA), autumn (September, October, November: SON), and winter (December, January, February: DJF). Source: Trenberth et al. (2007).
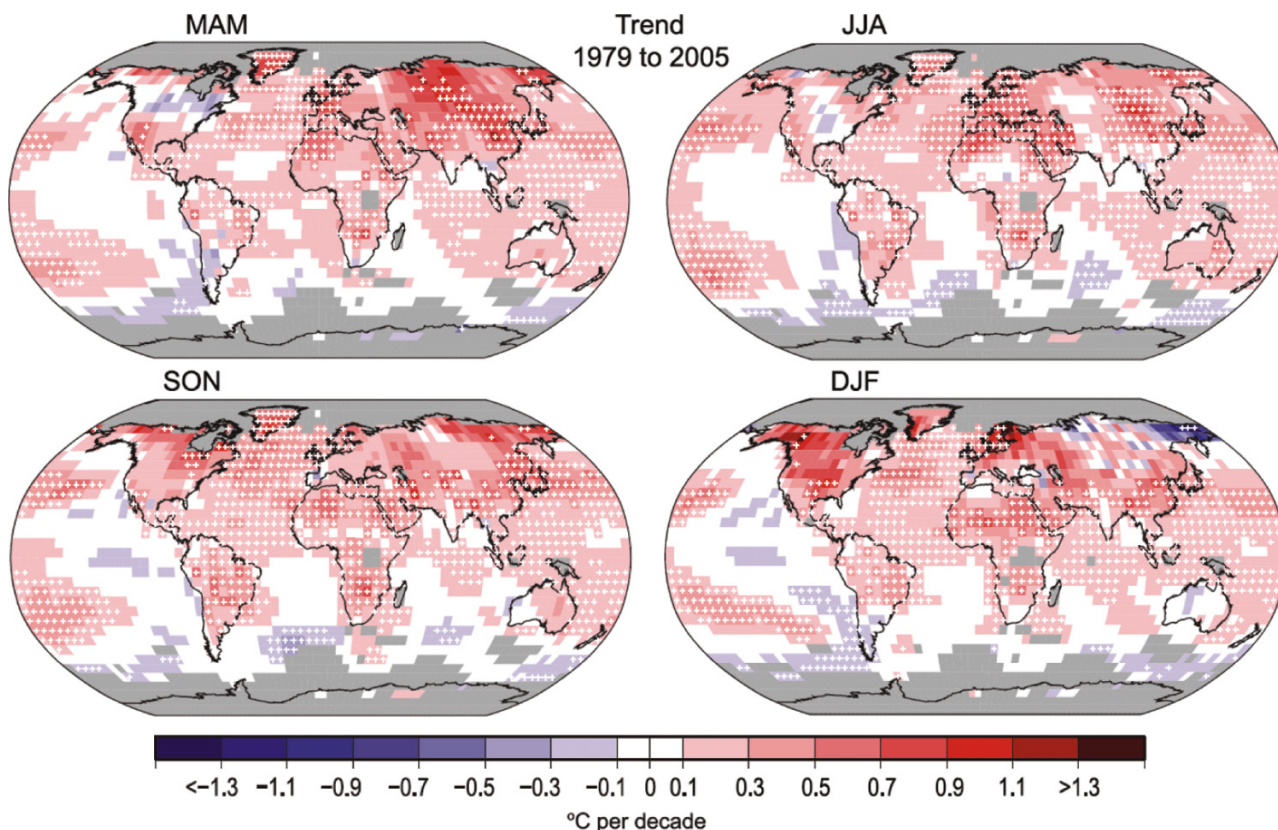
net evaporation are typically becoming more saline; lower SSS regions at high latitudes are typically becoming fresher (Fig. 17B; cf. Fig. 17C). In the North Atlantic, the northward transport of saline, subtropical waters by the Atlantic meridional and subsequent overturning circulation complicates this scenario slightly.

Since the 1979 advent of satellite measurement, Arctic sea-ice cover in both summer and winter has been in steady decline (Fig. 18). Minimum September sea-ice extent has decreased at a rate of 12.04% per decade, whereas March sea-ice extent has been decreasing at a rate of 2.74% per decade. Given this scenario, the extrapolated extension of the

Fig. 14. Blue line, average sea surface temperature (SST) over the Gulf of St. Lawrence from May to November; green line, annual mean average surface air temperature averaged over nine Environment Canada stations around the Gulf of St. Lawrence (Sept-Îles, Natasquan, Blanc-Sablon, Mont-Joli, Gaspé, Daniel's Harbour, Charlottetown, Îles de-la-Madeleine, and Port aux Basques); black line, annual mean average surface air temperature from Charlottetown together with the linear trend; red line, annual mean average surface air temperature from Pointe-au-Père (near Rimouski) together with the linear trend. Galbraith et al. (2010) used the surface air temperature records as a proxy for Gulf of St. Lawrence SST based on a very high correlation between these time series during modern times. Source: Galbraith et al. (2010).
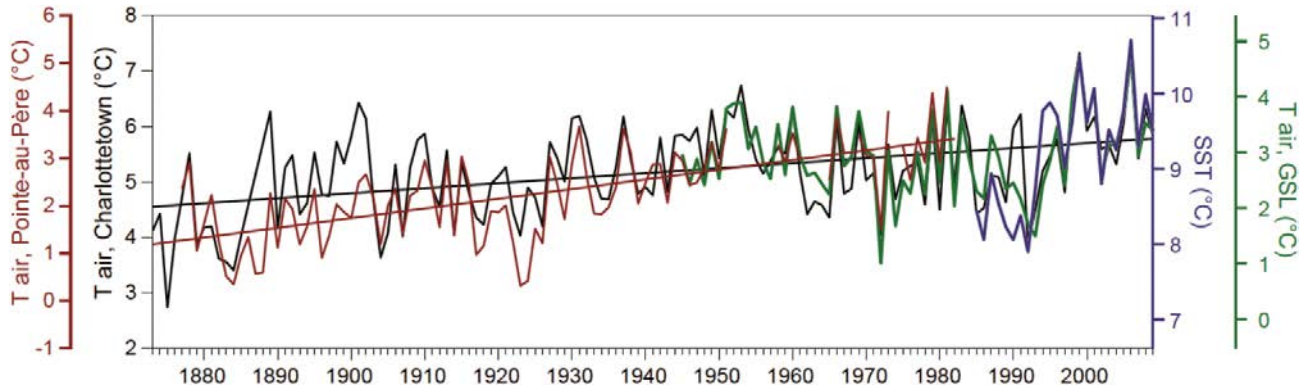
Fig. 15. Time series (1950–2005) of vertically averaged temperature (0–175 m) from Station 27 located on the Newfoundland Shelf off St. John's Harbour (47°32′N, 52°35.2′W). Source: Templeman (2010).
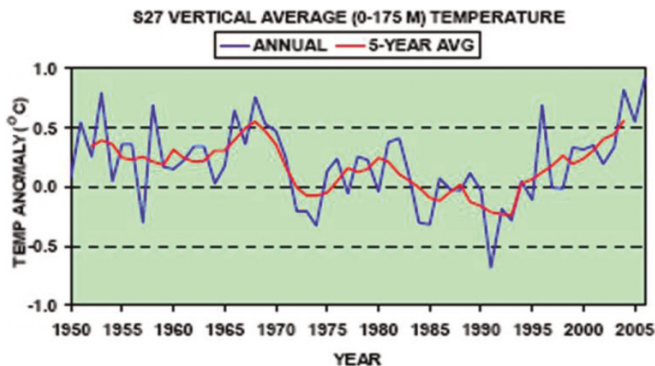
linear trend points to an ice-free Arctic in September 2071 (section 4.2.2).

In recent years, Arctic sea-ice thickness has also been decreasing. Kwok and Rothrock (2009) found that the average winter sea-ice thickness declined 48% from 3.60 m in 1980 to 1.89 m in 2008. Using a coupled ice-ocean model, Lindsay et al. (2009) further estimated that the September sea-ice thickness has, since 1987, been decreasing at a rate of 57 cm per decade. Given that older ice tends to be thicker than younger ice, this decline in thickness can be readily seen in the observed reduction of Arctic sea-ice age during the last 30 years (Fig. 19). Average ice age in the Arctic continued to decline in 2009 (Fig. 19C) before recovering slightly in 2010 (Figs. 19D, 20). While there was a slight aging of ice in 2011, less than 30% of the Arctic ice is more than a year old, a drop of nearly 50% since 1979 (Fig. 20).

Significant trends towards declining sea-ice extent and thickness are also evident on Canada's east coast. For example, the linear trend of sea-ice coverage from 1979 to 2011 is –3.9% (±2.2%) per decade for the Gulf of St. Lawrence region (top panel in Fig. 21) and –3.1% (±1.5%) per decade for the Labrador Sea region (bottom panel in Fig. 21). Sea-ice extent was at, or below, record minimum levels during the winters of both 2009/2010 and 2010/2011. During 2010/2011, sea-ice coverage (5.2%) was 72% below its 1979–2011 winter average in the Gulf of St. Lawrence. In the same year, the ice cover in the Labrador Sea region was at a record low of only 4.0% coverage, 71% below the 1979–2011 average.

#### 4.2.2. Projected changes during the 21st century

The retreat of September Arctic sea ice over the last several decades was much faster than that simulated by any of the climate models assessed in the IPCC's 4th Assessment Report (AR4; Stroeve et al. 2007). This is likely due to a combination of several model deficiencies: (i) incomplete representation of ice albedo (reflecting power) physics, including the treatment of melt ponds (e.g., Pedersen et al. 2009); (ii) omission of surface warming associated with the deposition of black carbon (soot) (e.g., Flanner et al. 2007; Ramanathan and Carmichael 2008); and (iii) incomplete representation of the physics of vertical and horizontal mixing in the ocean (e.g., Arzel et al. 2006).

Climate models assessed by the IPCC's AR4 revealed a linear relationship between annual mean Arctic sea-ice extent and global mean surface air temperature (NRC 2011; Winton 2011). In particular, annually averaged Arctic sea-ice extent is predicted to decrease by about 15% per degree of global warming (NRC 2011). However, Winton (2011) showed that the observational record exhibits a greater decline per degree of warming than the AR4 models. Several recent studies (Boé et al. 2009; Wang and Overland 2009; Zhang 2010) have used observational constraints on subsets of the AR4 climate model simulations to estimate when, during summer, the Arctic might first become ice-free. Both Wang and Overland (2009) and Zhang (2010) suggested a nearly ice-free summer could occur in the Arctic as early as the late 2030s, with a few remaining ice covered areas existing around the northern edge of the Canadian Arctic Archipelago (cf. Fig. 13).

The 5th phase of the Coupled Model Intercomparison Project (CMIP5) involves contributions from twenty modelling groups worldwide. Environment Canada's Canadian Centre

Fig. 16. Observed trends in annual precipitation (1979–2005) in % per decade (the % is relative to the 1961–1990 mean). Green, increasing precipitation trends; brown, decreasing precipitation; grey, insufficient data available to determine a trend reliably. Source: Trenberth et al. (2007).
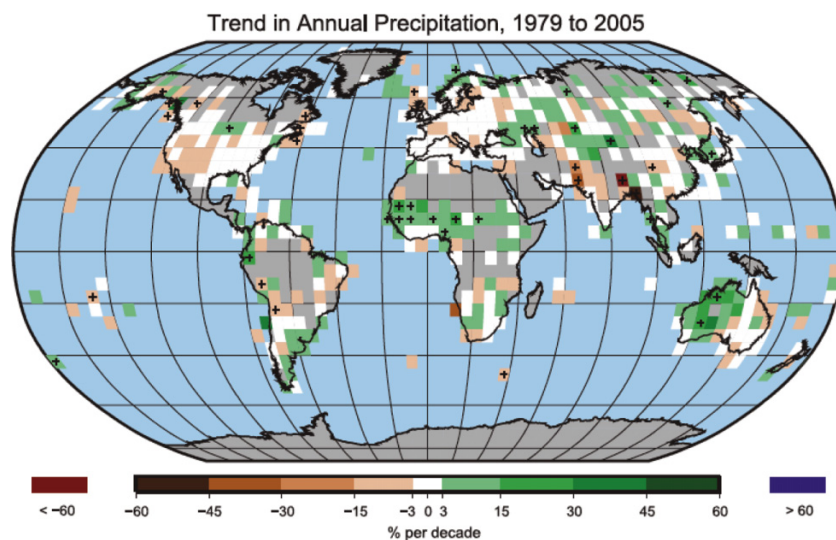
for Climate Modelling and Analysis provides the Canadian contribution. Future climate projections, coordinated through CMIP5, are also being used in the 5th Assessment Report of the Intergovernmental Panel on Climate Change (AR5). As part of CMIP5/AR5, four new Representative Concentration Pathways (RCPs) were developed to provide models with estimates of future radiative forcing associated with anthropogenic emissions of greenhouse gases and aerosols. Each of the components is named after the radiative forcing (in W m^{-2}; W, watts) it produces at year 2100 (Fig. 22). By definition, RCP4.5 yields a net radiative forcing of 4.5 W m^{-2} in year 2100.

Projected warming of globally averaged sea surface temperature (SST; Fig. 21) over the next two decades (~0.24 °C per decade in the Canadian CGCM4/CanCM4 model) is relatively insensitive to the emissions trajectory. However, projected outcomes diverge as the 21st century progresses. By year 2100, RCP4.5 yields 1.6 °C additional warming above the 1995–2005 average. The corresponding results for RCP 8.5 and RCP 2.6 are 2.6 and 0.9 °C, respectively.

The warming of the upper ocean is further illustrated in the zonally averaged potential temperature fields shown in Fig. 23. Subsurface warming is most pronounced where North Atlantic deep water forms in the northern hemisphere and where Antarctic Intermediate Water forms in the southern hemisphere. Projected regional warming patterns for the new RCP emissions trajectories are not yet available from the various modelling groups. Given this, and as an illustration of the projected regional change in surface temperature, we include Fig. 24 taken from Christensen et al. (2007).

Several features are notable and consistent with a continuation of existing trends. First, projected warming is amplified over land, relative to the oceans, because of the high heat capacity of water. Second, due to the downstream influence of the ocean, the warming is typically stronger in the middle of the continents, or on their leeward coasts, than on their windward coasts. Third, warming is greater at higher latitudes than at lower latitudes, due to local feedbacks, particularly ice/snow-albedo and ice-insulating feed-

backs. Moreover, at these same higher latitudes, warming is much greater in winter than in summer because this is when the feedbacks are strongest. The opposite is true over southern continental areas where reduced summer precipitation and soil moisture lead to an increased Bowen ratio (ratio of sensible to latent surface heat flux).

Over the middle to high latitudes of North America, including all of Canada, precipitation is expected to increase (Fig. 24). The opposite is true for the subtropical latitudes of the United States. In middle to high latitudes, the increase is greater in winter, with an increasing likelihood of rain instead of snow. At the same time, there is a greater likelihood of summer drought. Precipitation intensity and the interval between precipitation events are also projected to increase. That is, it will rain less often; but when it does rain, there will be more of it. As the difference between high latitude and subtropical temperatures shrinks, due to amplification of global warming at high latitudes, the overall number of mid-latitude storms is projected to decrease. Concurrently, the paths these storms take (storm tracks) will shift poleward and the likelihood of storms with intense wind speeds will increase. Therefore, there will be fewer overall storms; but when they do occur, there is a greater chance of their bringing stronger winds. The consequences for wave height and coastal erosion are potentially profound.

The projected increase in both high-latitude temperature and precipitation tends to make the high-latitude surface waters of the North Atlantic lighter and hence heightens their stability. As illustrated in Fig. 15 for the CGCM4/CanCM4 models, a slight weakening of the Atlantic meridional overturning circulation (AMOC) is predicted to occur during the 21st century. This is associated with a cessation of deep-water formation in the Labrador Sea and, hence, reduced vertical mixing of cold surface waters with underlying warmer subsurface waters. The net result is a reduction of the surface warming trend there (Fig. 24). Once the radiative forcing is stabilized, the AMOC recovers to its preindustrial level (Fig. 25). Gregory et al. (2005) found that, for all 11 models analyzed, the AMOC

Fig. 17. (A) Climatological annual mean surface salinity (in parts per thousand, on the right axis) from 1950–2000. (B) The linear trend in salinity overcalculated the period 1950–2008 (pss per 50 years; pss, practical salinity scale). (C) Freshwater flux ($m^3\ yr^{-1}$) at the ocean atmosphere interface averaged over the period 1980–1993. Source: Durack and Wijffels (2010).
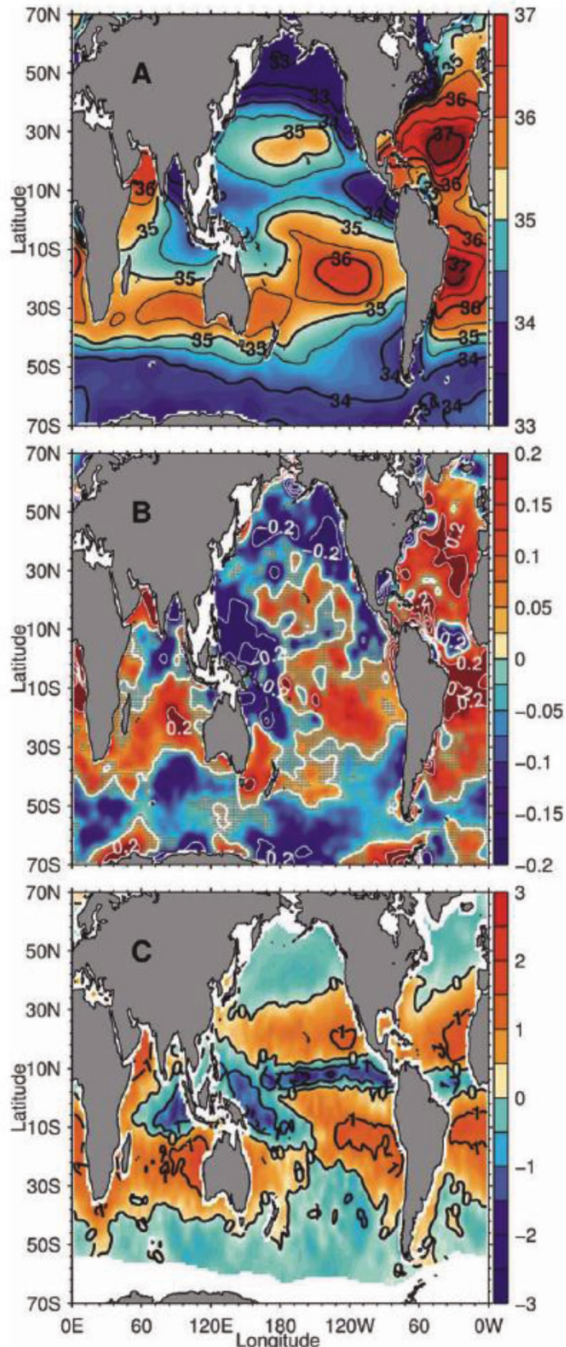
reduction was caused more by changes in surface heat flux than by alterations in surface freshwater flux.

As noted in Meehl et al. (2007), it is very unlikely that the AMOC will undergo an abrupt transition or collapse in the 21st century. As noted by Delworth et al. (2008), for such a phenomenon to occur, the sensitivity of the AMOC to forcing would have to be far greater than that seen in current models, or significant ablation of the Greenland ice sheet would be required, far exceeding even the most aggressive of current projections. Whereas neither possibility can be eliminated entirely, it is unlikely that the AMOC will collapse during the 21st century because of global warming

4.3. Observed and projected trends in regional wind systems

At the largest spatial scales, analyses of historical sea level pressure (SLP) and surface wind data have revealed that the intensity of both the Icelandic and Aleutian Low pressure cells, along with the strength of associated west-to-east blowing surface winds over the North Pacific and North Atlantic Oceans, increased between 1970 and 2005. Aspects of these increases are captured by indices tracking the state of atmospheric modes of variability over the North Pacific, North Atlantic, and Arctic oceans.

4.3.1. Aleutian Low

Major changes in the intensity of the Aleutian Low over the North Pacific were associated with the 1976–1977 climate shift, which saw a persistently more intense Aleutian Low, and a stronger counter-clockwise surface wind circulation in the decade after 1976, when compared with the previous decade (Trenberth 1990; Trenberth and Hurrell 1994). In the period between 1989 and 2011, the intensity of the Aleutian Low and the associated counter-clockwise winds over the North Pacific have, on average, been weaker than those in the 1980s, given the substantial year-to-year variability in the 1990–2011 period (Fig. 26).

Most models of climatic change project a northward displacement and strengthening of the mid-latitude west-to-east flow of winds which will be most pronounced in autumn and winter. These climate projections take into account a tendency for the centre of the Aleutian Low to move north of its historical climatological location. The reductions in surface pressure in the north are projected to be strongest in winter. The subtropical North Pacific High is also projected to intensify in summer, particularly off the coast of California and its Baja coast (Meehl et al. 2007).

4.3.2. Arctic Oscillation and Arctic Dipole

Interannual variability in sea level pressure over the northern hemisphere is dominated by changes in the Arctic Oscillation (AO). When the AO index is positive, westerly winds that form a polar vortex are intensified, with the core of the western winds displaced poleward. In the 20th century, the winter-averaged AO index exhibited persistent periods during which it remained in the same phase (i.e., either positive or negative). During the 1990s, the winter AO remained in the positive phase; whereas prior to 1970, the opposite prevailed. In the period between 1995 and 2010, the AO exhibited substantial year-to-year variability. In the summer of 2007, an unprecedented shift in atmospheric conditions occurred over the Arctic (Zhang et al. 2008). The typical tripole structure of the AO was replaced by what is now known as the Arctic Dipole which was also present in the late spring of 2009 and 2010 (Overland and Wang 2010). Many scenarios project decreases in Arctic surface pressure in the 21st century, as demonstrated by the average of scenarios from 13 different climate models (Fig. 27; Meehl et al. 2007). This contributes

Fig. 18. Arctic sea-ice extent from 1979 to the present for March and September. Area is expressed as an anomaly in million km² from the 1979–2000 average. The September 1979–2000 average is 6.04 million km² while the March 1979–2000 average is 15.75 million km². Source: National Snow and Ice Data Centre (NSIDC 2012).
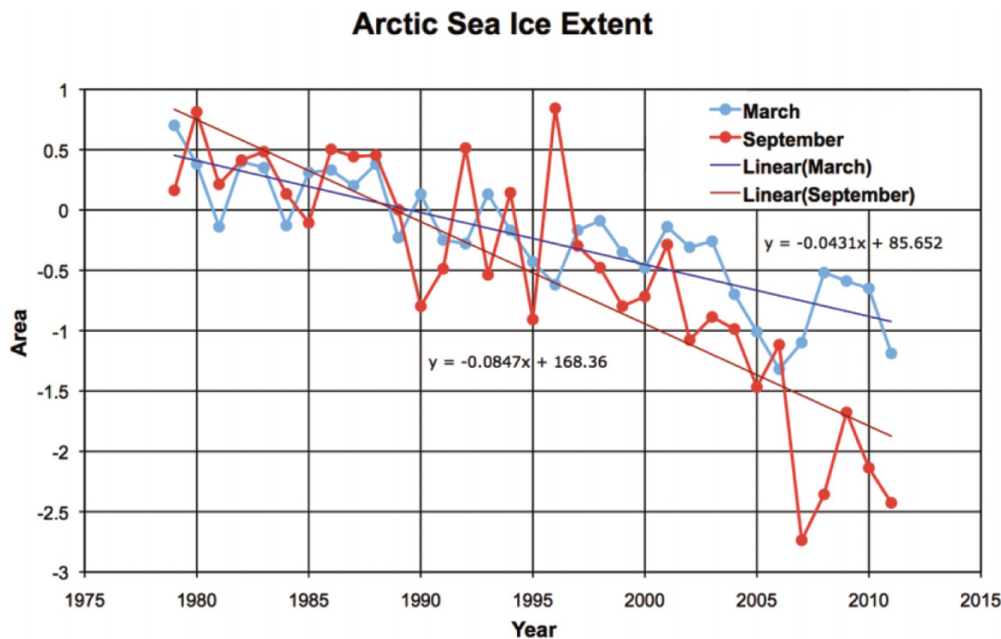
Fig. 19. Age of Arctic sea ice in March (*a*) 1988; (*b*) 2008; (*c*) 2009; (*d*) 2010. Red indicates ice of an age of five or more years. Source: Perovich et al. (2010).
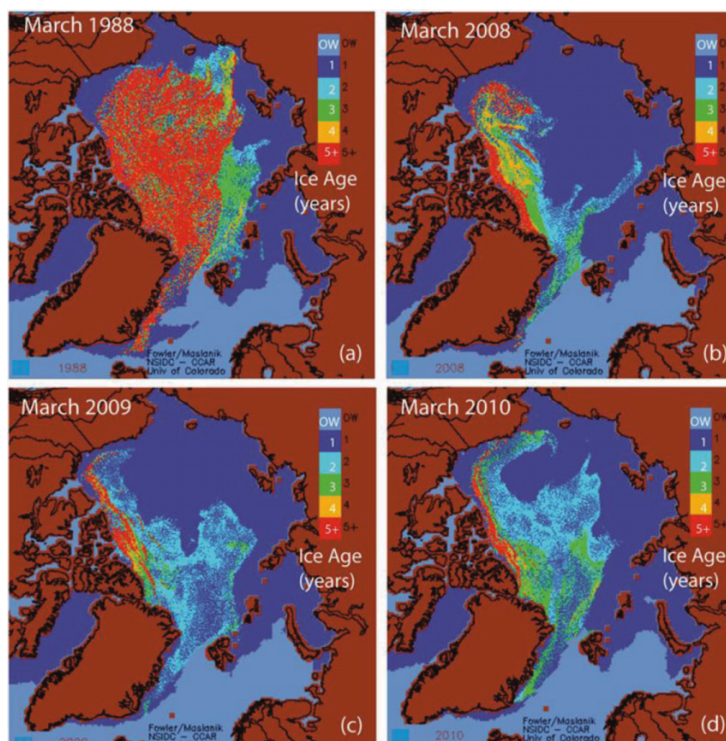

to an increase in indices of the AO and the North Atlantic Oscillation (NAO). However, the multimodel average from a larger number (*n* = 21) of models indicates that, because different climate model scenarios show alternative trends, the future behaviour of the AO (and NAO) index is highly uncertain.

4.3.3. Coastal upwelling and downwelling winds on Canada's west coast

In winter, Canada's west coast typically experiences intense wind-driven coastal downwelling; while in summer, winds are typically weaker and more variable. Off the west coast of

Fig. 20. Top: age of Arctic sea ice in March 2011 and the median ice age for March over the period 1979–2000. Bottom: percentage of March ice with a particular age from 1981 to 2011. Source: National Snow and Ice Data Centre (NSIDC 2012).

Arctic sea ice age at the end of winter
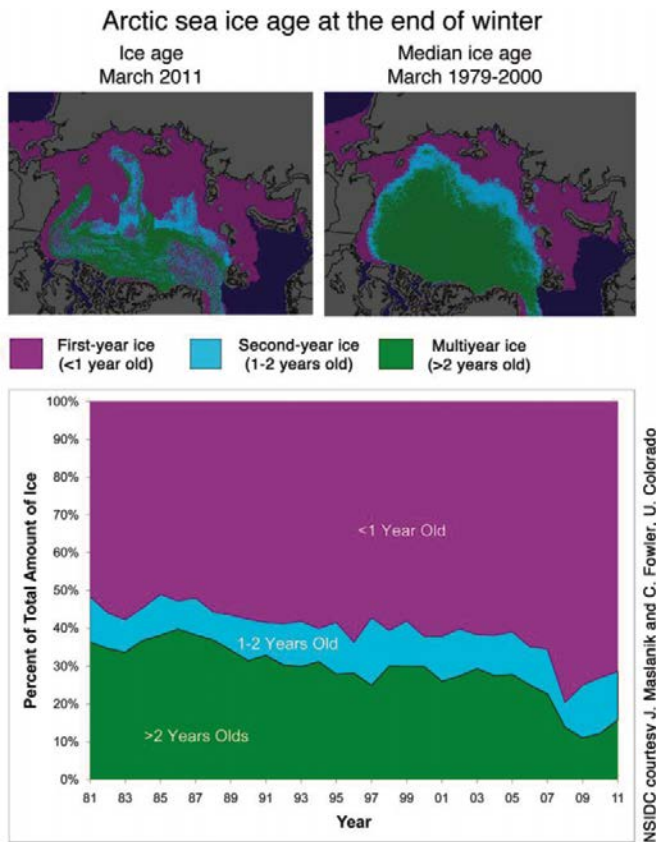
Vancouver Island, there are frequent periods of wind-driven coastal upwelling in summer. Coastal wind observations collected along this coast over the past 40 years do not indicate any clear trends in either summer upwelling or winter downwelling (Ianson and Flostrand 2010). Similarly, from 1948 to 2006, there are no definitive trends in local, wind-driven upwelling or winter downwelling for Queen Charlotte Sound and Hecate Strait. There are, however, especially large winter downwelling variations between years and decades (Cummins and Haigh 2010).

Merryfield et al. (2009) evaluated future trends in wind scenarios for locations off the west coast of Canada. Using 18 climate models, the researchers reported that upwelling winds will increase in speed by 5%–10% and rotate clockwise ~5°, a combination that leads to increased summertime upwelling, whereas ensemble mean changes in winter downwelling winds are not statistically significant.

4.4. Observed and projected trends in ENSO, PDO, and NPGO

4.4.1. El Niño Southern Oscillation (ENSO)

The behaviour of ENSO varied considerably from the late 19th to early 21st century, with especially active periods in the early 1900s and again since the 1950s. The period from 1925 to 1950 was relatively quiet. ENSO behaviour changed fol-

lowing the previously mentioned 1976–1977 climate shift; since then it has been characterized by a tendency for longer lasting and more intense ENSO events (Meehl et al. 2007). For example, the 1982–1983 and 1997–1998 El Niño events are the strongest on record. During the 1990s and through the early 2000s, a new type of El Niño was prominent, with maximal warming located near the International Dateline rather than the eastern equatorial Pacific (Yeh et al. 2009). This phenomenon has been recently named by various researchers as the central Pacific warming (CPW), dateline El Niño, the El Niño Modoki, and the warm pool El Niño. Among the climate model scenarios summarized in the IPCC's AR4, most projections indicate weak trends towards El Niño-like changes in the climate of the tropical Pacific, while only a few show weak trends toward La Niña-like conditions (Meehl et al. 2007). The models are more evenly split in projections for trends in the ratio of future variability of ENSO compared with that of past variability (Fig. 28).

Collins et al. (2010) use several metrics to screen future scenarios based on each climate model's ability to reproduce key characteristics of observed ENSO behaviour. The authors find that half the scenarios from the subset of "best"-performing models show increased ENSO variability, while the other half show decreased ENSO variability. Moreover, this same split between increased and decreased ENSO variability is present for the subset of "worst"-performing models. Recent analyses show that model projections of anthropogenic climate change are associated with an increased frequency of CPW El Niño events, when compared with the canonical eastern Pacific El Niño events that characterized most of the 20th Century (Yeh et al. 2009).

#### 4.4.2. Pacific Decadal Oscillation (PDO)

In records from 1900 to 2010, the PDO pattern varied across periods ranging from interannual to interdecadal, with a tendency for elevated variance at periods of 15-to-25 and 50-to-70 years, but with no distinct periodicities (Minobe 1999). Paleoclimate reconstructions for PDO behaviour over the past few centuries find sustained interannual to interdecadal variability across a range of timescales, with no predominant fixed bands of periodicities.

Overland and Wang (2007) evaluated several dozen 21st century scenarios produced by 18 climate models to better understand the behaviour of the PDO and North Pacific SST changes in a warming climate. The authors found that 10 of the climate models reproduce the spatial patterns and characteristic variability associated with the PDO in both 20th century "control" experiments as well as 21st century future scenarios. Nonetheless, the near-uniform warming trends become the most prominent pattern in the 21st century. By comparing multimodel average trends with individual model scenarios for SST in the central North Pacific, the authors demonstrate that, probably until some point in the 2030s to the 2050s, the anthropogenic climate change signal is likely to be swamped by natural-origin SST variations (some of which are associated with the PDO) (Fig. 29).

4.4.3. North Pacific Gyre Oscillation (NPGO)

Di Lorenzo et al. (2010) showed that decadal fluctuations in the NPGO are characterized by a pattern of SST anomalies that resemble the central Pacific warming (CPW) pattern of recent El Niño events and the tropical SST anomalies are responsible for an atmospheric teleconnection that forces the

Fig. 21. Top: percentage winter (November 26 through March 5) sea ice coverage in the Gulf of St. Lawrence (red area in top left corner of panel) for the period 1979–2011. Bottom: percentage winter (November 26 through March 5) sea ice coverage over the Labrador Sea region (red area in top left corner of panel) for the period 1979–2011. The linear trend is indicated in yellow. Source: Canadian Ice Service (CIS 2012).
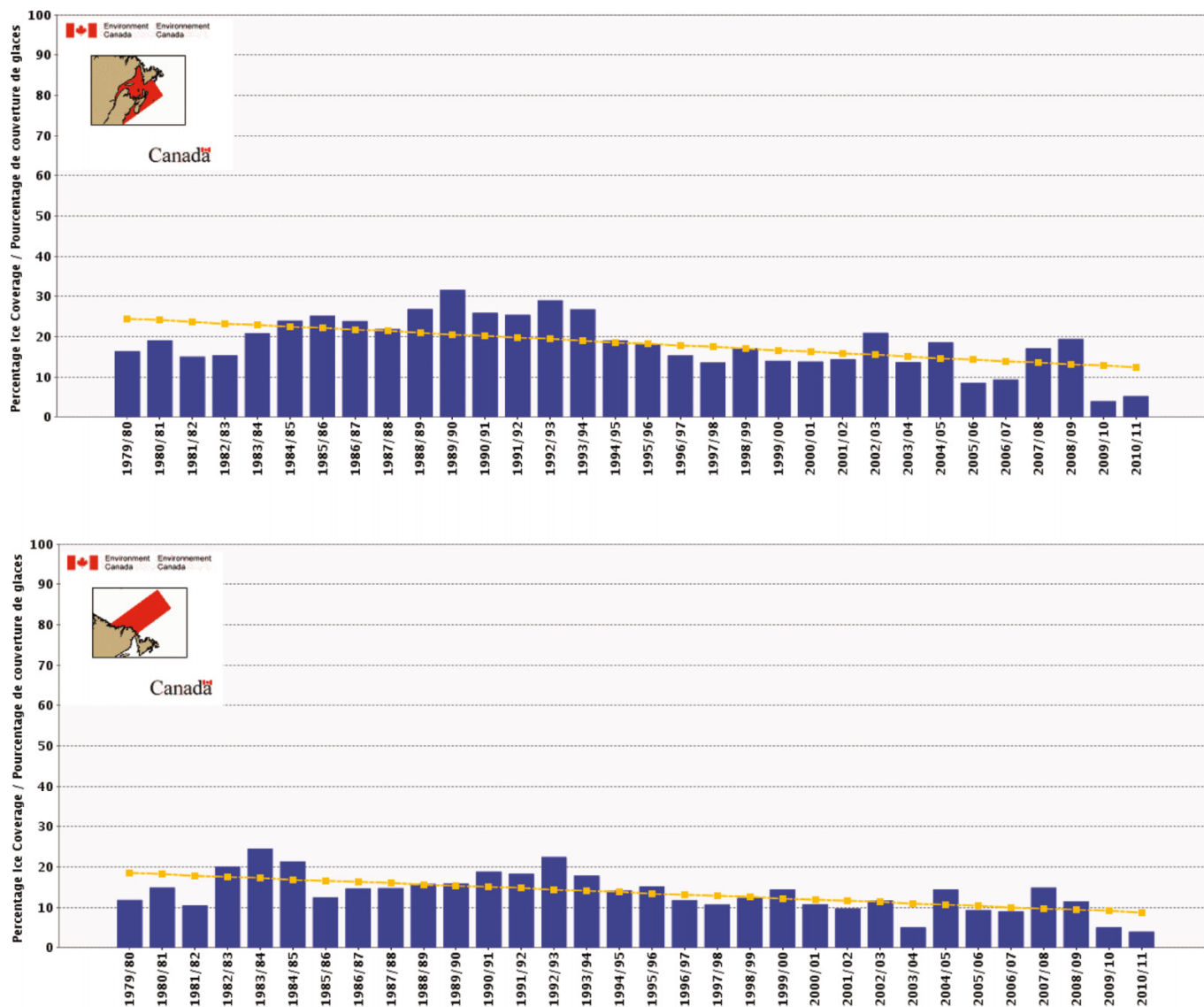
NPGO pattern in the North Pacific. If the 21st century climate scenarios indicating an increased frequency of the CPW El Niño events are realized, the NPGO is likely to play an increasingly predominant role in future North Pacific climate and oceanographic variability.

4.5. Coastal sea level

4.5.1. Present rate of change and coastal erosion

At the last glacial maximum, approximately 21 000 years ago, most of Canada was covered by the extensive Laurentide Ice Sheet. Whereas this ice sheet has long since melted, its effects are still evident on Canada's coastlines. When the Laurentide ice sheet grew, it depressed the surface of the earth below it. When the ice sheet melted, the crust began to return to its normal elevation. Today, the land is still rising in large parts of Canada, as a result of a process known as postglacial

or isostatic rebound (Fig. 30). Peltier (2004) estimated peak rates of isostatic uplift to be about 1.5 cm year^{-1} and occurring in the middle of Hudson Bay; isostatic uplift for the town of Churchill, Manitoba, is about 1.2 cm year^{-1}. Similarly, Lake Superior's northern shores are rising by 3–4 mm year^{-1}, thereby affecting its shorelines. Conversely, at the same time, the southern shore of Lake Michigan is sinking by about 1 mm year^{-1}. So, too, glacial isostatic adjustment along the coasts of BC and the Maritimes is causing the land to subside, which compounds the sea level rise associated with global warming. Halifax, St. John's, Victoria, and Richmond all have coastlines that are sinking by about 1 mm year^{-1}, solely as a consequence of the past melting of the Laurentide ice sheet. Tectonic compression coupled with subsequent uplift in the Cascadia subduction zone is causing the sea level to fall in parts of coastal BC.

Fig. 22. Projected ensemble average of the RCP 2.6, 4.5, and 8.5 globally averaged, annual mean SST from 1955 through to 2100. The output shown is from the Canadian CGCM4/CanCM4 contribution to CMIP5 (see Arora et al. 2011).
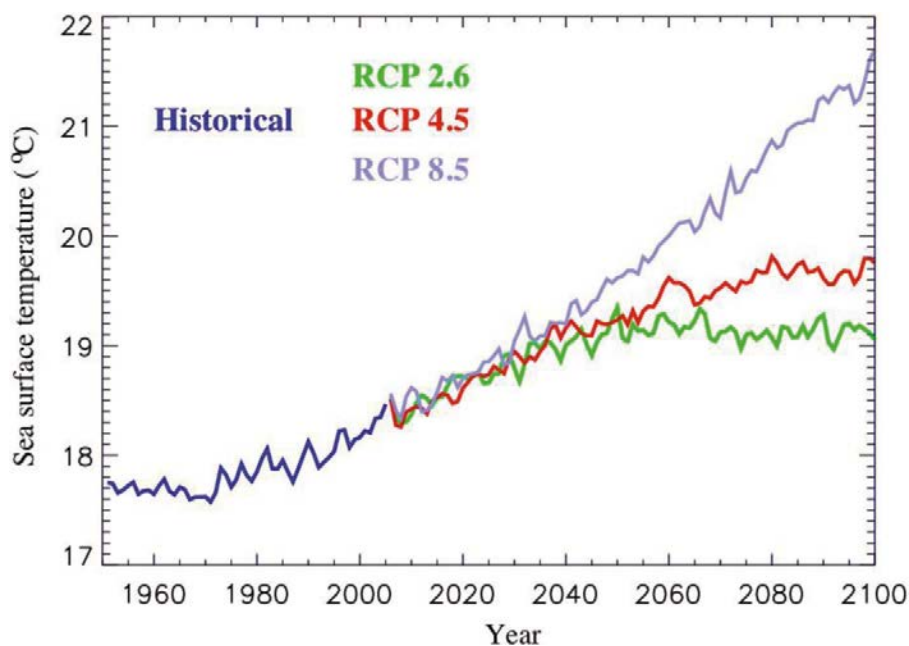
Sea level rise also occurs through thermal expansion when water temperatures increase. Another major component of sea level rise comes from the melting of glaciers and ice sheets on land. Observations reveal that, from 1993 to 2007, global sea levels rose by about 3.3 mm year^{-1} (Cazenave and Llovel 2010), with about 40% of this increase stemming from thermal expansion and approximately 60% originating from terrestrial ice melting. In recent years, the contribution from land ice melting has increased to 80% of the total sea level rise (Cazenave et al. 2009). As noted by Rignot et al. (2011), the combined loss of mass from the Greenland and Antarctic ice sheets has accelerated by 21.9 Gt year^{-2} (1 Gt = 1 billion tonnes) over the last 18 years.

The regional manifestation of global sea level rise is highly variable (Fig. 31), with patterns closely reflecting upper ocean temperature trends over the same period. From 1992 to 2009, sea level rise was great over much of the Atlantic, western Pacific, and Indian oceans, as well as in parts of the Arctic. In the eastern equatorial and North Pacific, sea level rise was small or even negative.

Enhanced inland flooding associated with storm surges and coastal erosion are potential impacts of sea level rise on Canadian coasts. Rising water tables, coupled with potential saltwater intrusion into wetlands and ground water, are also a concern for many coastal areas. Parts of Arctic Canada are particularly susceptible to coastal erosion, which is compounded by reduced ice cover (and hence enhanced coastal wave activity) and melting permafrost resulting from increasing temperatures (Fig. 32).

4.5.2. Projected changes during the 21st century

There have been substantial advances in scientific understanding of 21st century projected sea level rise since the release of the IPCC's AR4 (see Meehl et al. 2007). The IPCC report estimated 21st century global sea level rise of between 18 and 59 cm (relative to 1980–1999 average), depending on

the magnitude of future anthropogenic greenhouse gas emissions. The report further noted that the upper limit is uncertain, given that model-based estimates of sea level rise do not account for ice-sheet dynamics. More recent analyses have estimated a projected sea level increase from 1990 to 2100 of 75–190 cm (Vermeer and Rahmstorf 2009) or 80–200 cm (Pfeffer et al. 2008), with a central estimate of ~120 cm provided by Rahmstorf (2010).

Using 10 of the IPCC climate models, Yin et al. (2009) examined the regional response of projected sea level rise over the 20th century and concluded that the projected change in sea level was amplified in the Arctic and the northwest Atlantic, relative to the global mean. The amplification in the northwest Atlantic was a consequence of a weakening of the North Atlantic overturning. Shaw et al. (1998) undertook a comprehensive assessment of the vulnerability of the Canadian coast to sea level rise. Through a careful analysis of the contributions to regional sea level from tectonic and anthropogenic factors, they classified the Canadian coastline as being either at low, moderate, or high risk to sea level rise (Fig. 33).

The most at-risk regions are the coasts of Nova Scotia, New Brunswick, and Prince Edward Island, the entire Beaufort Sea coastline bordering Yukon and Northwest Territories, and the Fraser River Delta in BC. Much of the remaining BC coast was found to be at low risk to the effects of 21st century sea level rise, due to the nature of its high, rocky topography.

4.6. Water chemistry and stratification

Future climate change scenarios include both increased temperature and reduced salinity in the upper ocean at high latitudes. The combination of warmer and fresher surface waters leads to reduced density in the surface layer and increased stratification of the water column. By itself, increased vertical stratification is likely to reduce the flux of nutrients from deeper, denser waters

Fig. 23. Projected ensemble average of the RCP 2.6, 4.5, and 8.5 zonal mean potential temperature change from preindustrial times averaged over 2040–2060 and 2080–2100. Model results are shown from the CGCM4/CanCM4 contribution to CMIP5.
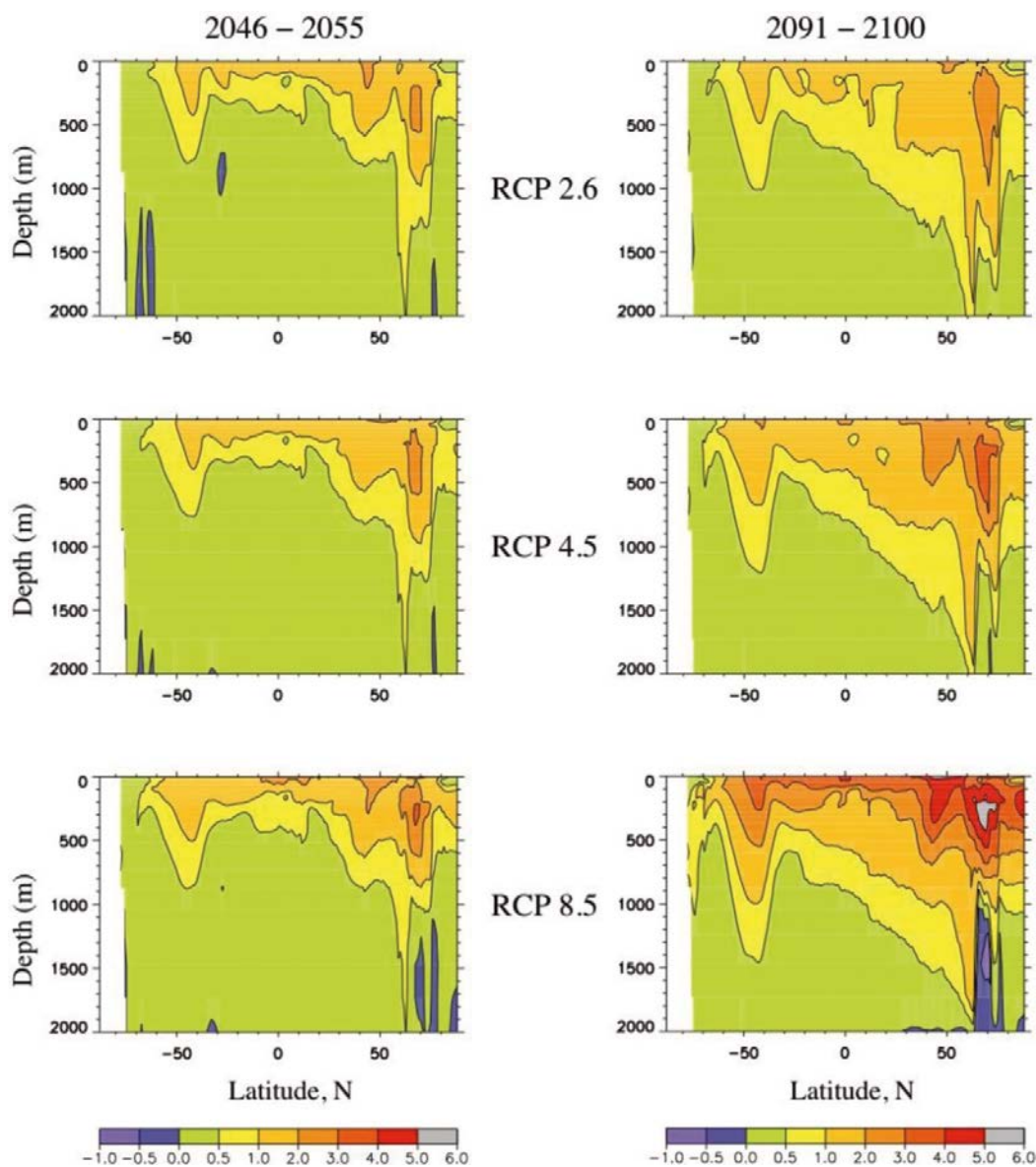

below the pycnocline upward to the surface layer and to reduce the ventilation of the thermocline. A reduced nutrient supply is likely to reduce phytoplankton production in Canada's nutrient-limited shelf ecosystems but to increase plankton productivity where nutrients are abundant and the availability of sunlight is limiting. Reduced ventilation of the thermocline is also expected to reduce the dissolved oxygen concentration at depth. This combination has already been observed in recent decades in the North Pacific Ocean (see below). Increased uptake of atmospheric CO_2 into the upper ocean has also led to increased ocean acidification. These trends are expected to continue beyond the 21st century (see below).

4.6.1. Observed trends in carbonate chemistry and pH

Rapid increases in the atmospheric concentration of CO_2 increase the ocean's uptake of CO_2, causing the oceans to become more acidic. Recent calculations estimate that approximately 26% of anthropogenic CO_2 emissions between 2000 and 2009 entered the global oceans (Friedlingstein et al. 2010). Between 1750 and 1994, the estimated uptake of anthropogenic carbon was calculated to have reduced the pH of the global surface ocean from 8.2 to 8.1, corresponding to a 30% increase in the H^+ concentration (Sabine et al. 2004). At the same time, the carbonate ion concentration in seawater has decreased dramatically, reducing the availability of calcium carbonates used by marine organisms to build hard shells and carbonate skeletal structures. Aragonite is one of the most soluble calcium carbonate minerals and is frequently incorporated into the hard parts of calcifying organisms. Shoaling of the aragonite saturation depth (the depth at which carbonate minerals dissolve more readily than they can form) has been
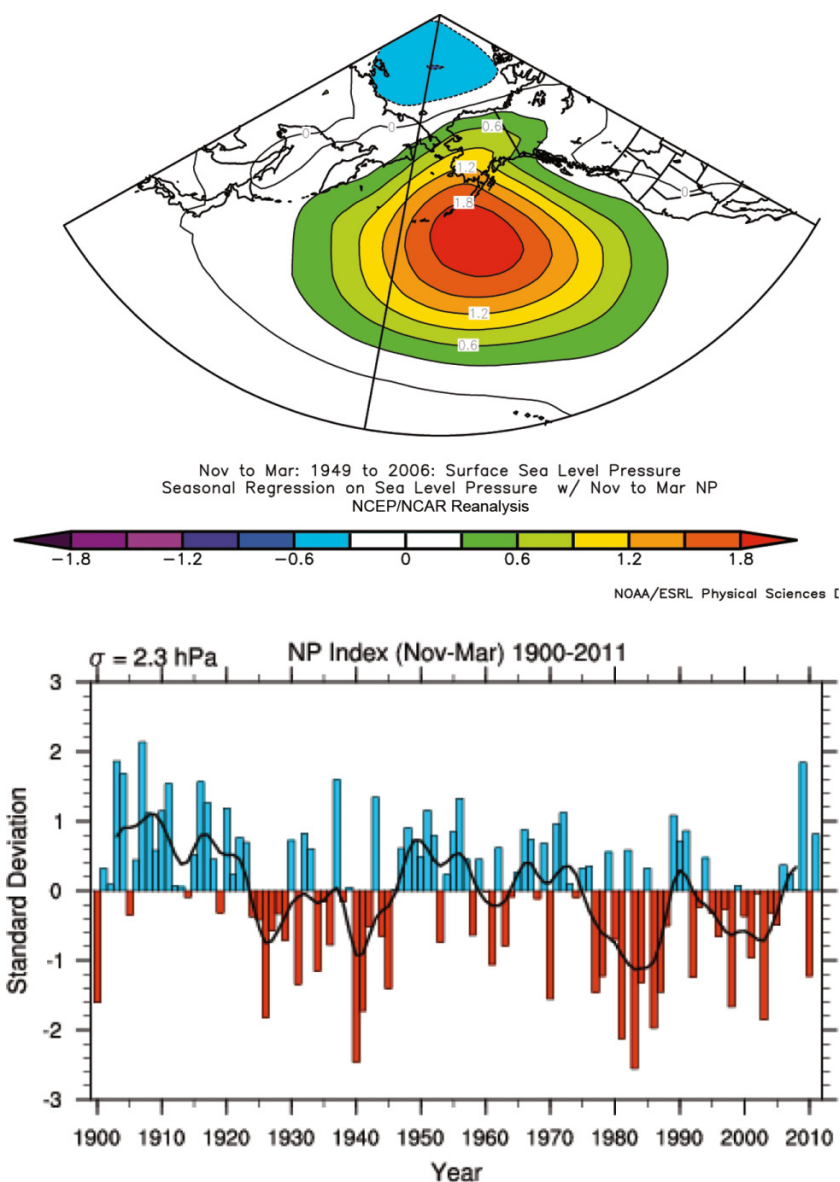
Fig. 24. Projected multimodel ensemble temperature (top) and precipitation (bottom) changes over North America from the 1980–1999 average to the average over 2080–2099. The mid-range A1B emissions scenario was used. The first column shows the annual mean response whereas the second and third columns provide the average winter (December–February, DJF) and summer (June–August, JJA) responses, respectively. Source: Christensen et al. (2007).
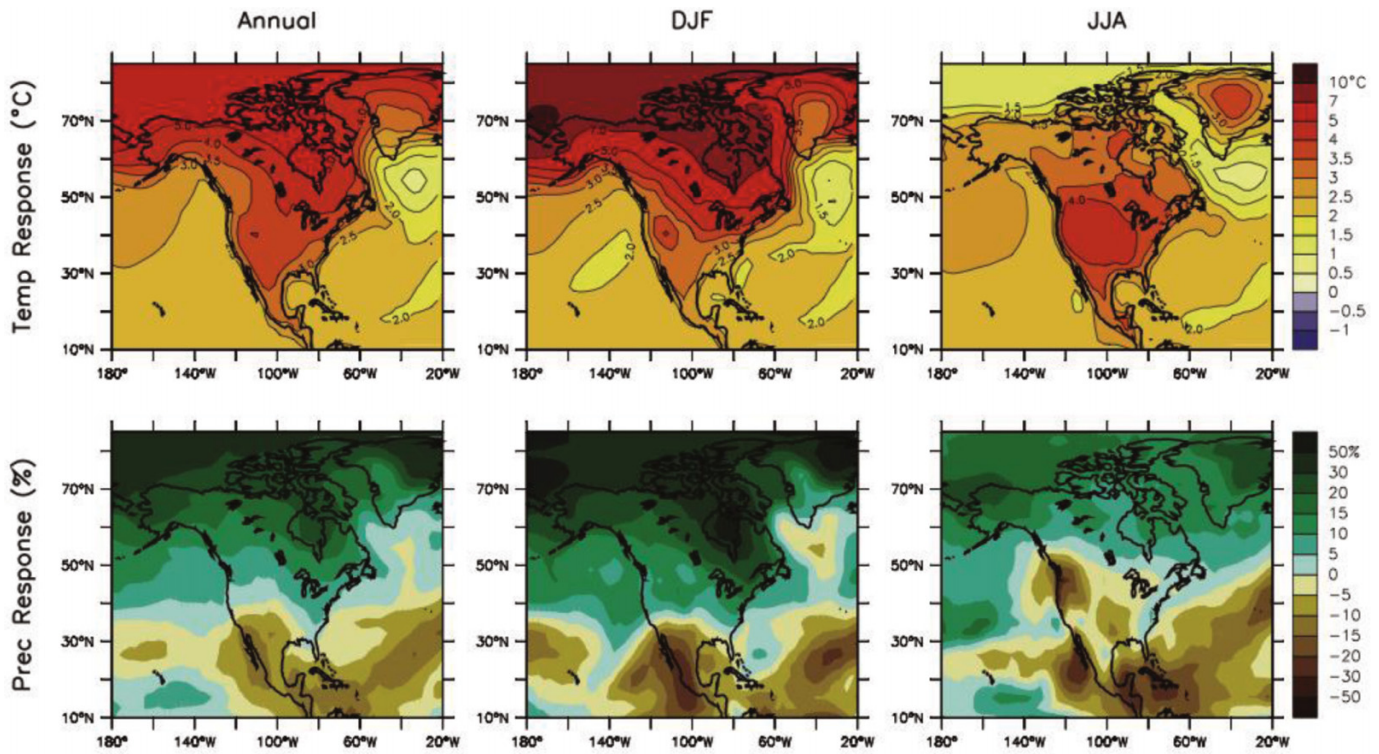
Fig. 25. Ensemble average outflow of the Atlantic meridional overturning circulation (AMOC) at 25°–30°S from 1850–2100. The RCP 4.5 extension to 2300 with radiative forcing held fixed at 4.5 W/m² after 2100 is also presented. Model results are shown from the CGCM4/CanCM4 contribution to CMIP5.
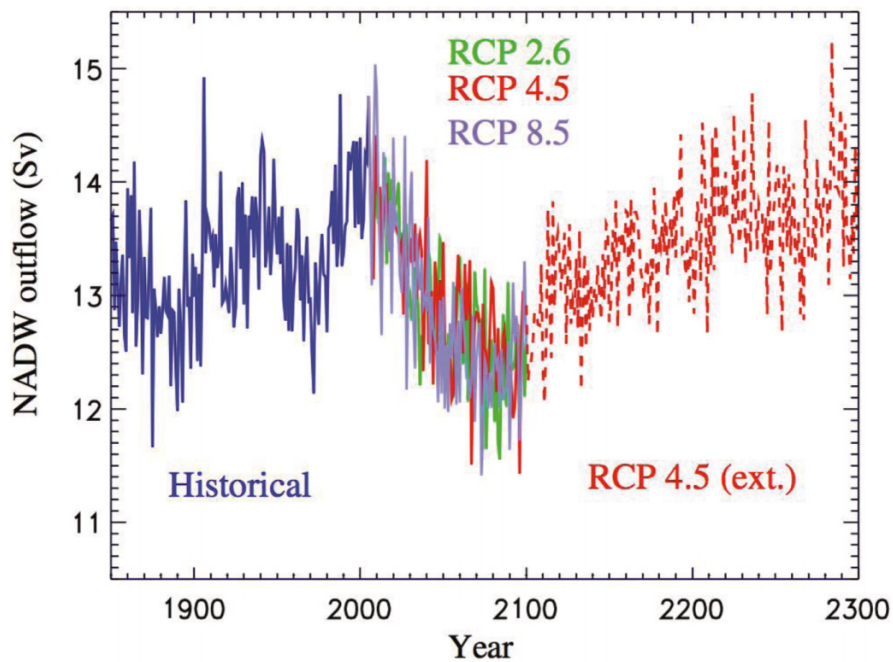
Fig. 26. Top: spatial pattern of Aleutian Low sea level pressure variations tracked by the "North Pacific" (NP) index of Trenberth and Hurrell (1994) (image created using NOAA's online plotting tool at http://www.esrl.noaa.gov/psd/data/correlation/. Bottom: time series of the Nov–Mar NP index from 1900 to 2011 (figure obtained from http://climatedataguide.ucar.edu/guidance/north-pacific-index-npi-trenberth-and-hurrell-monthly-and-winter; accessed 15-5-2011).

observed in all of the world's ocean basins. Future declines in carbonate saturation states are expected to have consequences for high-latitude marine ecosystems, primarily because baseline carbonate saturation states are relatively low in Subarctic and Arctic seas, including the North Pacific, Arctic, and North Atlantic oceans. Already, in 2008, surface waters in the Canada Basin of the Arctic Ocean were observed to be undersaturated with respect to aragonite. Recent trends towards undersaturated surface waters in the Canadian Arctic are linked to melting sea ice and increased upwelling of carbon-rich waters onto the continental shelf (Yamamoto-Kawai et al. 2009).

In the coastal waters of northern BC, saturation depths for aragonite are now approximately 300 m from the surface

(Cummins and Haigh 2010). Due to the ocean's uptake of anthropogenic CO_2 and consequent increases in acidification, this saturation depth is estimated to have shoaled by 50–200 m over the past century (Feely et al. 2008). The nearshore waters off the west coast of Vancouver Island are, in summer, frequently further undersaturated during those periods when coastal upwelling brings deep carbon-rich waters onto the shelf. Corrosive waters were actually observed at the surface near parts of the Oregon and California coast during a 2007 summer research cruise (Feely et al. 2008).

In the Gulf of St. Lawrence, recent findings reveal a significant pH decrease in hypoxic waters. In the 1930s, the in situ pH at >200 m depth in the lower St. Lawrence Estuary was about 7.90 (Fig. 34). Today, in situ pH levels are down to

Fig. 27. Multimodel mean of the regression of the leading (EOF) empirical orthogonal function of ensemble mean Northern Hemisphere sea level pressure (SLP; thin red line) relative to a 1900–1970 reference period with zero mean from 13 different climate models. The thick red line is a 10-year low-pass filtered version of the mean. The grey shading represents the intermodel spread at the 95% confidence level and is filtered. A filtered version of the observed sea level pressure from the Hadley Centre (HadSLP1) is shown in black. The regression coefficient for the winter following a major tropical eruption is marked by red, blue, and black triangles for the multimodel mean, the individual model mean, and observations, respectively. NH, Northern Hemisphere; hPa, hectopascal. Source: Meehl et al. (2007).
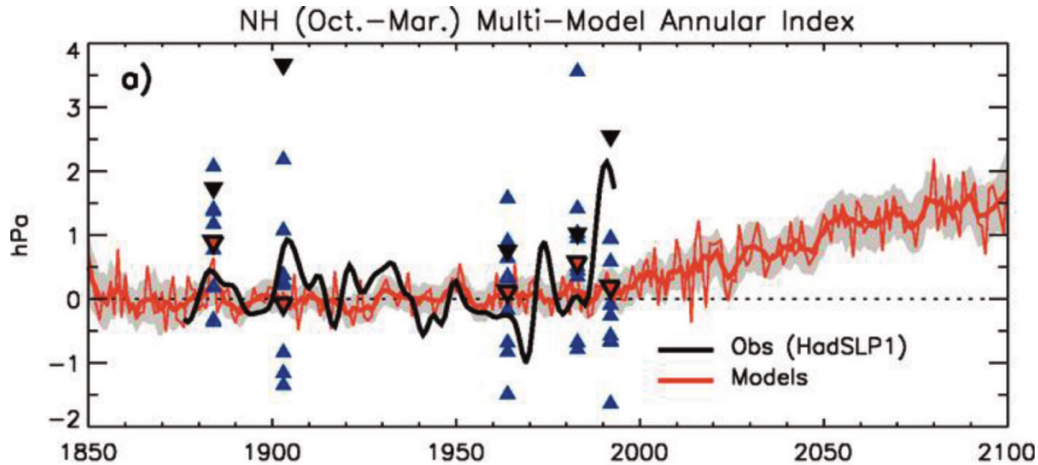
Fig. 28. Base state change in average tropical Pacific SSTs and change in El Niño variability simulated by climate models contributing to the IPCC's AR4. Source: Meehl et al. (2007).
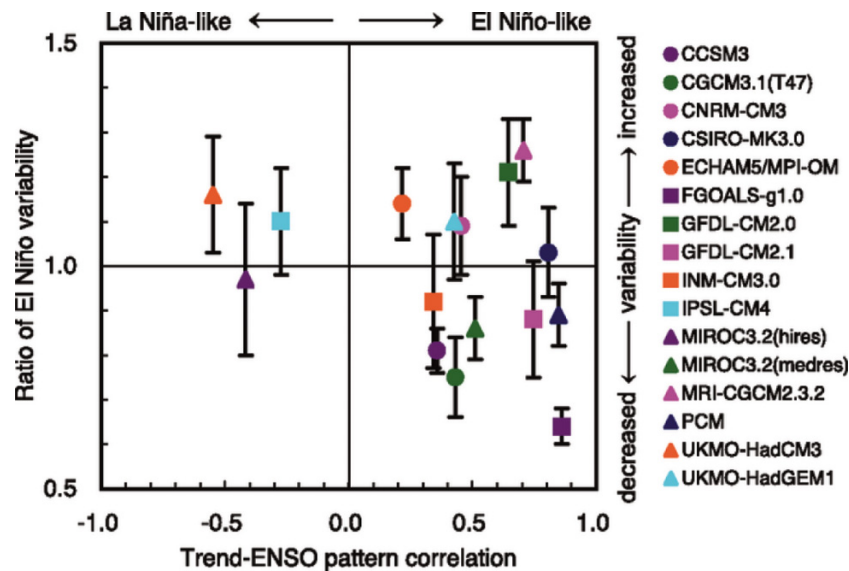

about 7.65, with some observations as low as 7.60. This change represents a 60%–90% increase in H^+ ions. In addition, the pH levels in the Lower St. Lawrence Estuary hypoxic waters have already reached levels expected for the surface ocean's global average for the end of the 21st century. These findings suggest that an increased flux of organic matter to bottom waters in the St. Lawrence may have increased respiration and resulted in the region's lower pH conditions (Dufour et al. 2010).

Climate system models predict extremely rapid declines in ocean pH (i.e., increases in ocean acidity) in the next century under a wide range of future greenhouse gas emissions scenarios (Orr et al. 2005). Multimodel projections based on

scenarios considered in the IPCC's AR4 give reductions in pH of between 0.14 and 0.35 units in the 21st century, adding to the already documented decrease of 0.10 units from preindustrial times (Bindoff et al. 2007).

Cooley et al. (2012) used simulations with a climate system model to identify the "transition decade" wherein future aragonite saturation states become distinctly different from those simulated for 2010. For Ocean Station PAPA (located in the Gulf of Alaska; Fig. 7), this transition is predicted to occur in the 2030s (Fig. 35). In fact, this modeling study suggests that most of Canada's coastal waters will experience a transition to a new envelope of aragonite saturation states during the 2030s (Fig. 36).

Fig. 29. Projected winter SST anomalies (°C) relative to a 1980–1999 base period for the central North Pacific Ocean (in the centre of the PDO SST pattern). Thin grey lines indicate individual ensemble member projections from 10 different climate models under the A1B greenhouse gas emissions scenario, while coloured curves are the ensemble means from four of the individual models. The bold blue curve indicates the trend of the all-model ensemble mean. Source: Overland and Wang (2007).
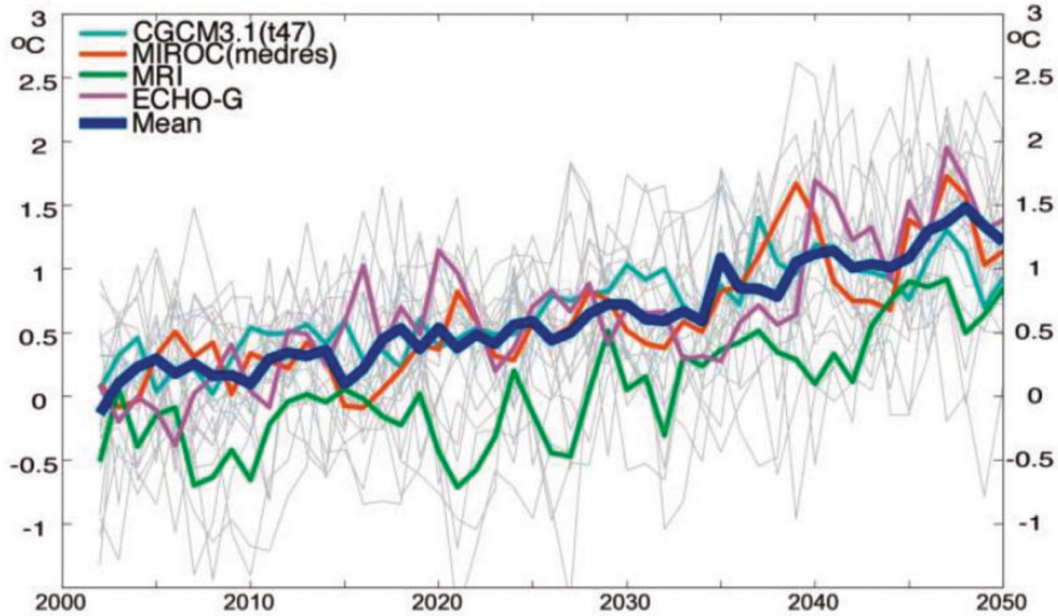
Fig. 30. Present-day vertical uplift in mm year^{-1} of the Earth's crust in North America, as projected by the ICE-5G model of Peltier (2004).
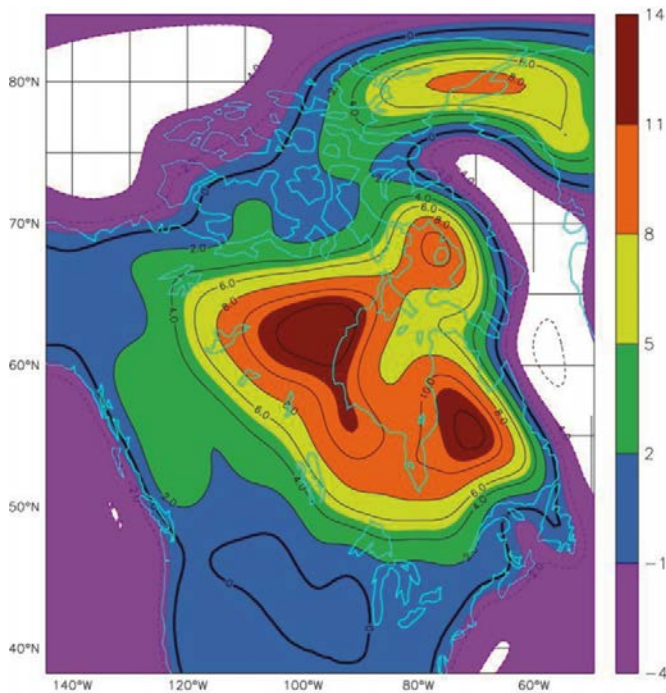
4.6.2. Dissolved oxygen

Over the past 50 years, trends and variations in dissolved oxygen have been documented for many parts of the world's upper oceans. The available data are insufficient to indicate if the changes in O_2 are caused by natural variability or are trends that are likely to persist in the future. However, the data indicate that large-scale changes in ocean physics do influence natural biogeochemical cycles, and thus the cycles of O_2 and CO_2 are likely to undergo changes if ocean circulation changes persist (Bindoff et al. 2007).

Differences in dissolved oxygen concentrations between the late 1990s and mid-1980s on two transects across the North Pacific Ocean reveal a pattern of increase in the upper 100 m and of decline at depths between 100 and 400 m (Fig. 37). A time series of observations from Ocean Station PAPA also shows declining oxygen concentrations from depths of 100 and 400 m between 1956 and 2006 (Whitney et al. 2007). Figure 38 presents trends in oxygen concentration for waters below 100 m depth along the North American coast from southern California to Haida Gwaii (Crawford and Irvine 2009). These trends are based on time series data of at least 25 years duration. Declines in dissolved oxygen are seen at all depths below the mixed layer and along the entire coast. The greatest declines are found within the 200–300 m depth range. Within this range, the rate of decline represents about 1% of the dissolved oxygen per year in BC coastal waters. The decline is attributed to the weakening of the ventilation of surface waters off the coast of Asia, a trend linked to freshening and warming that increases the water's stratification (Whitney et al. 2007).

In the 1930s and early 1970s, oxygen levels in deep waters of the Gulf of St. Lawrence Estuary were above the hypoxic threshold of 30% saturation. The deep waters of the estuary were briefly hypoxic in the early 1960s and have consistently been hypoxic at about 19%–21% saturation since 1984. One half to two thirds of this decrease is associated with changes in source water masses at the continental shelf (Gilbert et al. 2005).

Fig. 31. Regional trends in sea level rise over the period October 1992 – July 2009, as derived from satellite altimetry. Source: Nicholls and Cazenave (2010).
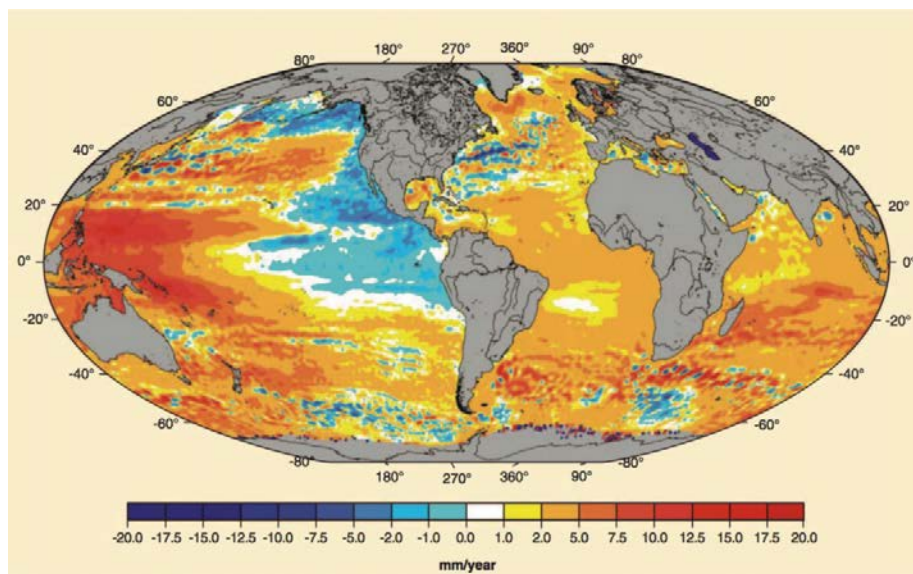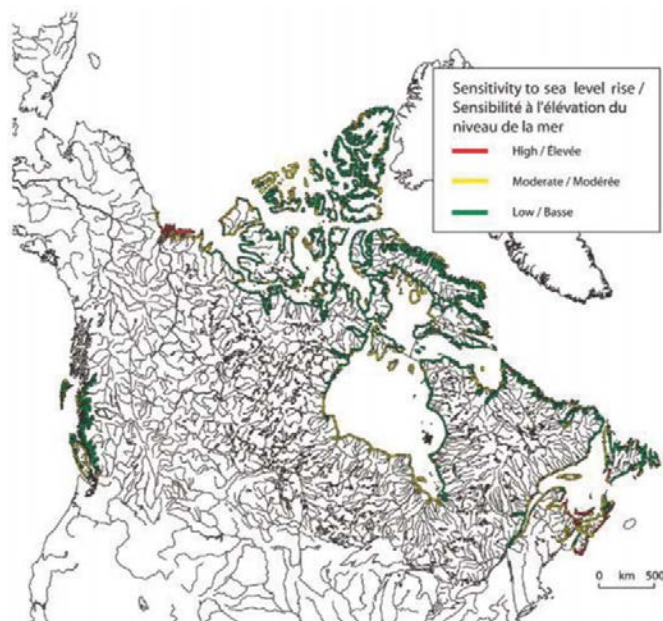

Fig. 32. Photograph taken in 2004 showing coastal erosion of exposed permafrost at Drew Point in Alaska. Source: http://energy.usgs.gov/alaska/ak_coastalerosion_images.html (accessed 6–5–2011).

Fig. 33. Sensitivity of Canada's marine coastline to sea level rise as assessed by Shaw et al. (1998). Available from Warren et al. 2004.

In offshore areas of the Newfoundland and Labrador coasts, there is significant mixing of highly oxygenated cold water, and therefore hypoxia is generally not a major consideration. However, there have been reports of low oxygen levels along northeastern Newfoundland, where levels in the 1990s were the lowest in a 70-year time series (Kiceniuk and Colbourne 1997). While hypoxia is not currently reported to be a significant issue within this region, occurring mainly in small fjords with restricted circulation and soft organic bottoms, it is likely that many Newfoundland and Labrador harbours could experience hypoxic conditions at some times of the year due to organic loading from nearby fish plants or other forms of

development (e.g., mines, lumber mills, sewage, and agricultural runoff) (Templeman 2010).

4.6.3. Nutrient pools

Essential plant nutrients in the ocean's surface layer are typically consumed rapidly in the process of photosynthesis by marine algae. In high-latitude systems, a spring phytoplankton bloom is typically initiated when daylight is sufficient to both warm the upper ocean enough to stratify the surface layer and supply light needed for photosynthesis. In the absence of a nutrient supply, photosynthesis by phyto-

Fig. 34. Observed pH between 200 and 320 m depth in the Lower St. Lawrence Estuary. Source: Dufour et al. (2010).
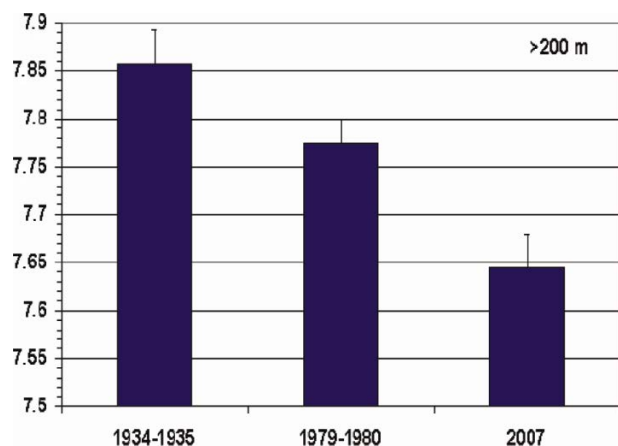
plankton quickly depletes nutrients in the sunlit upper ocean. Below the sunlit portion of the upper ocean, nutrients are typically abundant year-round. Thus, the concentration of surface nutrients can be influenced by surface mixing, given that a reduction in mixing leads to a decreased supply and concentration of surface nutrients. In most of the Pacific Ocean, the observed surface warming and freshening trends between 1950 and 2005 act in the same direction and contribute to reduced mixing (Bindoff et al. 2007). This is consistent with regional observations in the northeast Pacific Ocean (e.g., Freeland et al. 1997).

Time series data combining surface nitrate and silicate concentrations in shelf and slope waters off southern Vancouver Island from 1988 to 2008 indicate substantial seasonal, interannual, and decadal variability (Fig. 39). Petrie and Yeats (2000) documented interannual variations in nitrate, silicate, and phosphate at a depth of 150 m in the Gulf of Maine and on the central Scotian Shelf that were only weakly correlated between these two regions in the 1960s through 1990s period. They suggested that these variations were related to changes in water mass structure. As in most marine ecosystems, nitrogen availability is the limiting factor to primary production in Gulf of Maine and Scotian Shelf waters. Although there is significant seasonal variation in nitrate concentrations in surface and deep waters of the Gulf of Maine and Scotian Shelf, no long-term trends have been observed at coastal and offshore sampling stations in the last 10 years (Worcester and Parker 2010).

Time series data on the primary marine nutrients do not appear to be available for the northern coastal waters of BC or the Canadian Arctic (Cummins and Haigh 2010; Niemi et al. 2010).

4.6.4. Stratification

Whitney and Freeland (1999) argued that, over the latter portion of the 20th century, increased stratification in the northeast Pacific (due mainly to basin freshening) reduced the flux of nutrients across the pycnocline and into the surface layer. In contrast, for most of the Atlantic, temperature and salinity trends in the 1950–2005 period generally acted in opposite directions and had mixed effects on upper ocean stratification (Bindoff et al. 2007).

On the Scotian Shelf, the average 0–50 m stratification index increased between 1960 and 2008, but most significantly in the 1990s. From the mid to late 1990s, the index was at or

near its maximum over the 59-year record (Fig. 40; Petrie et al. 2009). Important changes in stratification are also noted in the eastern Gulf of Maine and Georges Bank, coupled with increasing temperature and changes in salinity. Stratification increased steadily from the mid-1980s on Georges Bank and in the eastern Gulf of Maine (Fig. 41; Worcester and Parker 2010).

5. Trends in Canadian marine biodiversity

5.1. Introduction

Trends in biodiversity are driven by human and environmental pressures. The primary purpose of this section is to address the question in the terms of reference, "What are the past and current trends and associated uncertainties in Canadian marine biodiversity?", focussing on trends affected by fishing and climate change. As clarified earlier (section 2), the section deals primarily with population trends (species data are presented where they exist) because of the lack of trend data for most species (section 3). An additional consideration is that many, but not all, species for which estimates of abundance can be obtained are, or were, of commercial importance. Among those for which trend data exist, the time periods often extend several decades. This is true for many fishes. Data are available for phytoplankton and zooplankton for periods of time extending, in some areas, to 50 years, although shorter periods are much more common. Abundance data are available for some marine mammals, although not always on an annual basis, with the exception of some intensively monitored species, such as Pacific killer whales (*Orcinus orca*) and Atlantic pinnipeds (e.g., harp seals, *Pagophilus groenlandicus*). For seabirds, long-term data exist for colonies on all three coasts, rendering this one of the few taxonomic groups (in addition to marine mammals) for which trend data have been collected in the Arctic. Data are limited for macroinvertebrates.
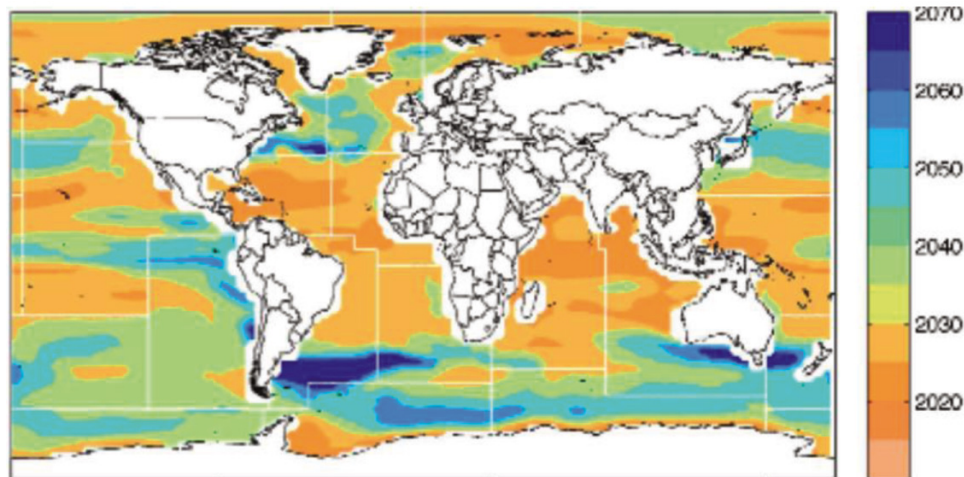
5.2. Marine species at risk

In the absence of yearly abundance data for many (indeed most) species, one means of evaluating the directional change in marine biodiversity over the past half century is to examine the numbers of marine species assessed as being at risk in Canada by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC). A legislatively recognized body, independent from government, COSEWIC is responsible for assessing the status of species believed to be at heightened chance of extinction, communicating its assessments to society and various levels of jurisdictional authority, and advising the federal government of those species that warrant inclusion on the national legal list of species at risk and, thus, some form of protection under the *Species at Risk Act* (SARA). The list of species assessed by COSEWIC to be at risk differs from the national SARA legal list of species at risk. The former is based solely on the best available information pertaining to a species' risk of extinction, whereas the decision to include a species on the SARA list is influenced by nonscientific influences, such as the perceived political and socio-economic consequences of a listing decision. In addition to making assessments at the species level, the Act provides for the assessment of populations, or groups of populations, below the species level, acknowledging implicitly that such designatable units (DUs) represent irreplaceable units of biodiversity critical to the persistence of biological species. COSEWIC identifies discreteness and evolutionary significance as the primary criteria for recognizing DUs (COSEWIC 2011b). Given its definition in SARA, the legislatively

Fig. 35. Time series of monthly mean CCSM3-modeled surface aragonite saturation state for Ocean Station PAPA (50°N, 145°W), with the 10-year running average (red) shown for reference. At Station PAPA, the normal range of annual variability (area between the black lines), or "envelope", will no longer overlap that of 2010 (area between the light blue lines) in approximately 2031. Source: Cooley et al. (2012).
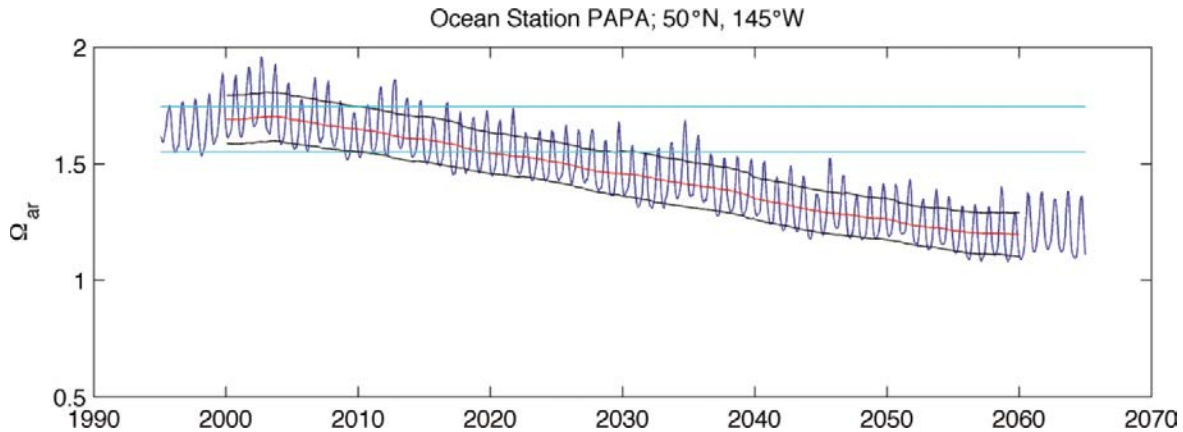
Fig. 36. Transition decades when future surface aragonite saturation states will no longer overlap those of 2010. From Cooley et al. (2012).

defined "wildlife species" can be considered functionally equivalent to a DU.

As of January 2012, COSEWIC had assessed 112 marine wildlife species (including diadromous fishes) as being either extinct ($n = 3$) or a species at risk (i.e., wildlife species assessed as extirpated (no longer in existence in the wild in Canada), endangered, threatened or special concern) (Table 2). The extinct species include sea mink (*Neovision macrodon*; a mammal), great auk (*Pinguinus impennis*; a bird), and eelgrass limpet (*Lottia alveus*; a mollusc). Marine ($n = 41$) and diadromous fishes ($n = 27$) comprise slightly more than half of the wildlife species assessed as endangered, threatened or special concern. Marine mammals ($n = 35$) have the 2nd highest representation in these three assessment categories. Seabirds ($n = 8$), molluscs ($n = 3$), and reptiles ($n = 3$, all of which are sea turtles) comprise the remaining taxonomic categories. Of the total of 640 species assessed by COSEWIC as being at risk (as of January 2012), approximately 20% are marine.

Based on quantitative and qualitative inspection of temporal trends in species status assessments (e.g., Hutchings and Festa-Bianchet 2009; COSEWIC Species Specialist Subcommittee

Annual Reports), and accounting for the number of species that have been assessed relative to the numbers present in Canada (e.g., Table 1), the Panel draws the following conclusions on future trends in the assessment of marine species. The greatest increase in numbers of marine species at risk will almost certainly be experienced by diadromous fishes, primarily because of the anticipated increased focus by COSEWIC on Pacific salmon (COSEWIC Marine Fishes Species Specialist Subcommittee 2010–2011 Annual Report). It is highly probable that the number of diadromous fishes at risk will increase from their current 27 to more than 50 (perhaps to as many as 70 or 80) in the coming decade because of the high number of Pacific salmon DUs forecasted to be at heightened risk of extinction. Primarily because of data limitations, the numbers of wholly marine fishes at risk might not exceed 50 or 60 in total (41 had been assessed as being at risk as of January 2012). It is also unlikely that the number of species at risk among marine mammals, seabirds, reptiles, and molluscs will increase appreciably, if at all, in the coming decade.

Fig. 37. Changes in oxygen concentration (μmol kg^{-1}) along two sections in the North Pacific (bottom panel). Top left panel: difference (1999–1985) along 47°N. Top right panel: difference (1997–1984) at 152°W. Blue colours indicate a decrease and yellow colours indicate an increase in oxygen over time. The differences were calculated using density as the vertical coordinate. Source: Bindoff et al. (2007).
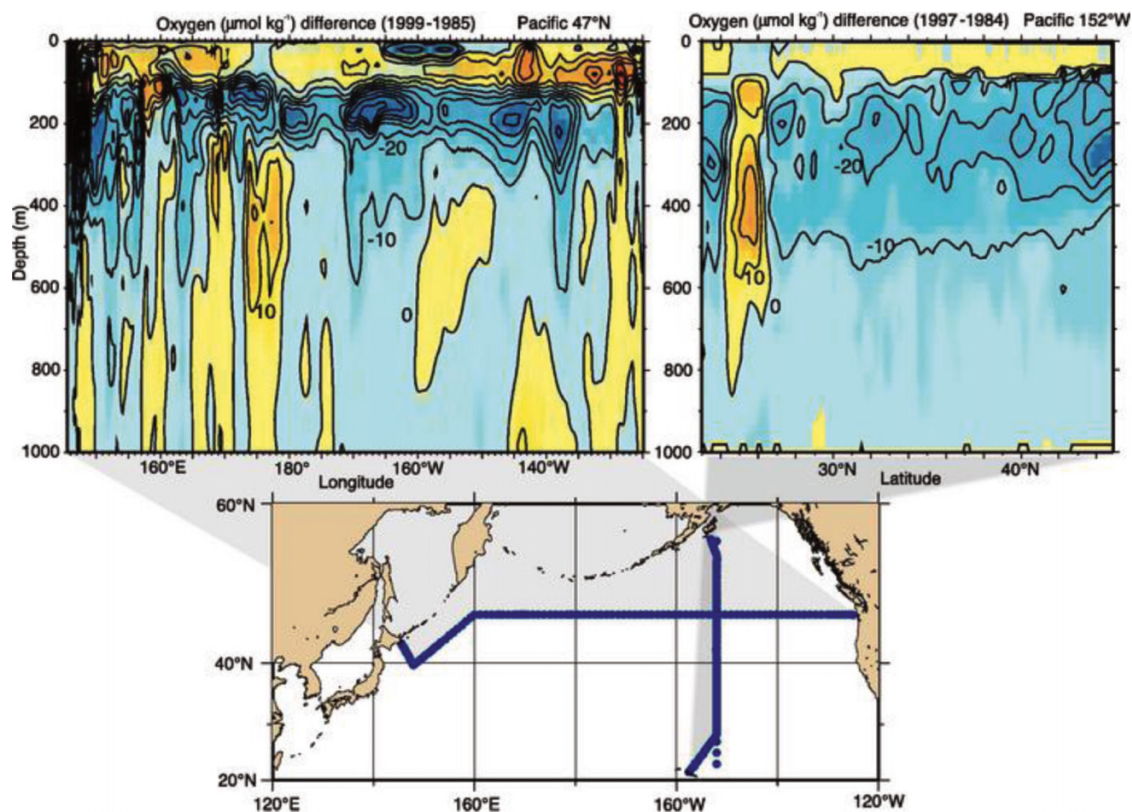
Fig. 38. Trends in oxygen concentration for waters below 100 m depth along the North American coast from southern California (SCA) to Haida Gwaii (WCQCI) (DFO 2009). These trends are based on time series data of at least 25 years duration. Source: Crawford and Irvine (2009).
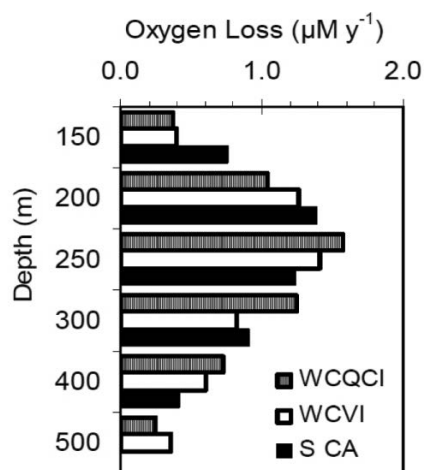
### 5.3. Metrics of population status

It is not uncommon to use rate or magnitude of decline in abundance as metrics of increased chance of harm to a population or species. For example, this is the basis for one of the extinction-risk criteria developed by the International Union for Conservation of Nature (IUCN; and modified for use by COSEWIC) to assess the status of species believed to be at heightened chance of extinction. Globally, and based on abundance estimates available from fish stock assessments, marine fishes declined 38% from the period 1970–1974 to the period 2002–2006; the trend is similar for both pelagic and demersal species (Hutchings et al. 2010). From one perspective, this magnitude of reduction over a period of nearly 40 years would be viewed as extremely problematic. The IUCN, for example, uses a decline rate of 30% (experienced over the longer of 10 years or three generations) as a threshold above which species are assessed as Vulnerable ("threatened", using COSEWIC's terminology). However, based on fisheries production models, if a species or a population had declined by 30% from a virgin or unfished state (B_0; something that can be estimated from models, such as the relationship between stock size and recruitment), it would not be considered to be at heightened risk of extinction; rather, it would be predicted that the population would be approaching a level of abundance at which the maximum sustainable yield (MSY) for the population would be obtained. Depending on the stock-recruitment model used, the stock biomass at which MSY is obtained (B_{MSY}) is estimated to range between 25% and 40% of B_0 (Hilborn and Stokes 2010). In other words, reductions of 60% to 75% would result in a population attaining B_{MSY}.

Thus, among species for which declines can be attributable solely to exploitation (and not, for example, habitat change), it

Fig. 39. Surface nitrate and silicate concentrations in shelf (here defined as the continental shelf and just beyond the shelf break) and slope waters (here defined as the region beyond the 1000 m depth contour between the slope and Alaskan Gyre) off southern Vancouver Island from May 1988 to August 2008. Source: Ianson and Flostrand (2010).
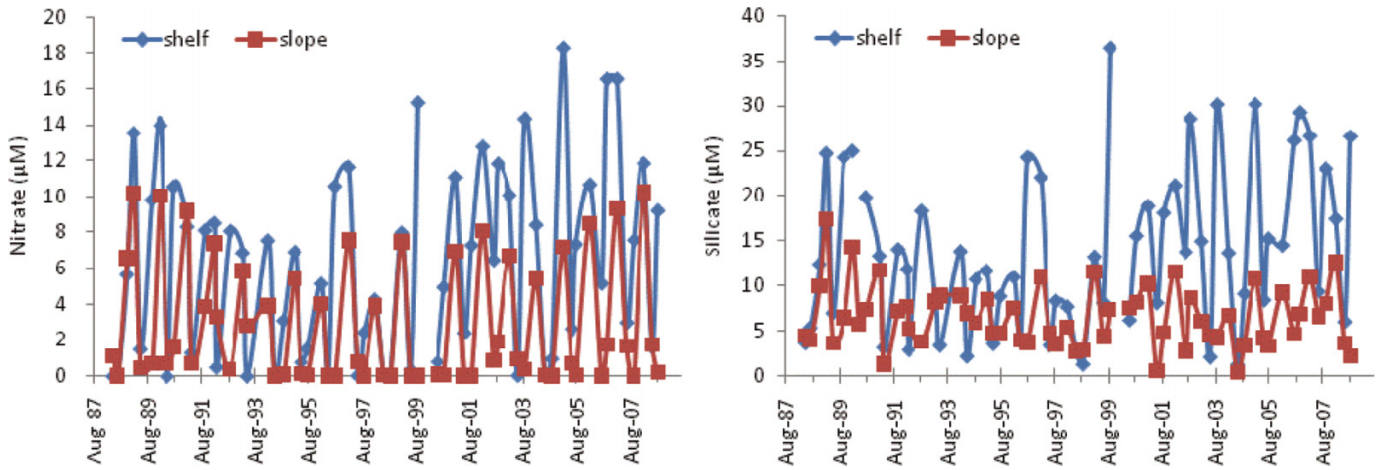
Fig. 40. The mean annual (dashed line) and 5-year running mean (solid line) of the stratification index over the Scotian Shelf. Standard error estimates for each annual value are shown. Source: Petrie et al. (2009).
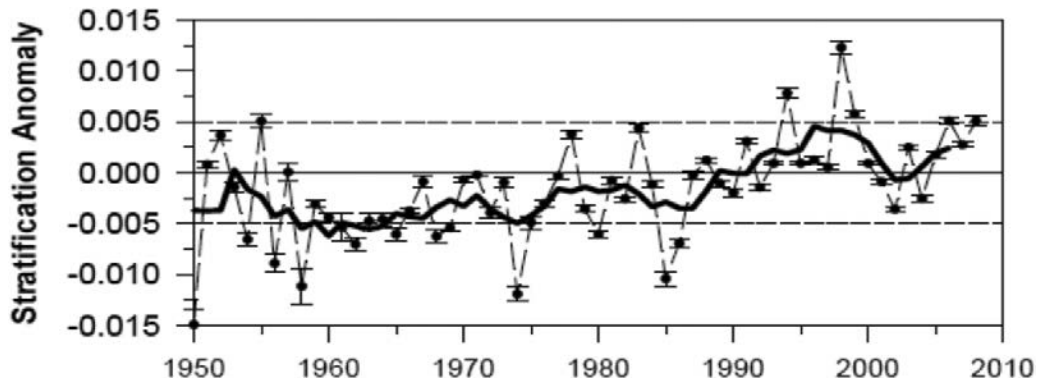
Fig. 41. Trends in stratification for the eastern Gulf of Maine and Georges Bank. Source: Worcester and Parker (2010).
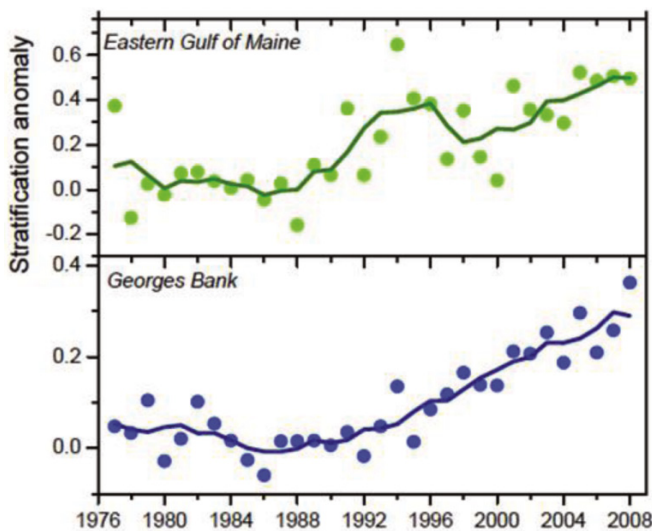
might be appropriate to interpret reductions in abundance relative to some point of reference of species or population productivity, such as a "target reference point" or a "limit reference point". A target reference point (TRP) identifies the long-term objective for a fishery in terms of population biomass (B), whereas a limit reference point (LRP) represents a low population biomass that should be avoided. These TRPs and LRPs are often expressed as a depletion measure, i.e., some fraction of B_{MSY} or B_0.

Although Canada has not established TRPs for most of its commercially exploited marine fishes, particularly those on the east coast (Hutchings et al. 2012a), many jurisdictions, including the US, New Zealand, and parts of the European Union, have adopted B_{MSY} as their TRP in harvesting management plans (the TRP in Australia is higher at 1.2 B_{MSY}). The Pacific Fisheries Management Council in the US uses a default value of B_{MSY} of 0.4B_0, which would correspond to a 60% decline from a population's unfished state (Hilborn and Stokes 2010). LRPs, which are also used extensively in the countries that have adopted TRPs, are commonly identified as 0.5B_{MSY}. DFO has established reference points for the harvesting of marine mammals (www.dfo-mpo.gc.ca/science/

Table 2. Marine wildlife species assessments by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC), as of January 2012.

Extinct	Extirpated	Endangered	Threatened	Special concern
Marine Mammals				
Sea Mink (Pacific)	Grey Whale (Atlantic)	Beluga Whale (Eastern Hudson Bay; Ungava Bay)	Beluga Whale (St. Lawrence Estuary; Cumberland Sound)	Beluga Whale (Eastern High Arctic – Baffin Bay; Western Hudson Bay)
		Blue Whale (Atlantic; Pacific)	Northern Fur Seal (Pacific)	Narwhal (Arctic)
		Killer Whale (Pacific Southern Resident)	Killer Whale (Pacific: 3DUs)	Sea Otter (Pacific)
		North Atlantic Right Whale	Fin Whale (Pacific)	Harbour Porpoise (Atlantic; Pacific)
		North Pacific Right Whale		Steller Sea Lion (Pacific)
		Northern Bottlenose Whale (Scotian Shelf)		Atlantic Walrus
		Sei Whale (Pacific)		Polar Bear (Arctic)
				Bowhead Whale (Beaufort Sea; Eastern Canada)
				Fin Whale (Atlantic)
				Grey Whale (NW Atlantic; Eastern Pacific)
				Killer Whale (NW Atlantic – Eastern Arctic)
				Humpback Whale (Pacific)
				Northern Bottlenose Whale (Davis Strait)
				Sowerby's Beaked whale (Atlantic)
Marine Fishes				
		Atlantic Cod (4 DUs)	Bocaccio (Pacific)	Atlantic Cod (Lakes on Baffin Island)
		Roundnose Grenadier (Atlantic)	Cusk (Atlantic)	Spiny Dogfish (Atlantic)
		Porbeagle (Atlantic)	Shortfin Mako (Atlantic)	Roughhead Grenadier (Atlantic)
		Deepwater Redfish (Gulf St. Lawrence – Laurentian Channel)	Deepwater Redfish (Northern Population)	Rougheye Rockfish (Pacific: Type I; Type II)
		Basking Shark (Pacific)	American Plaice (Newfoundland and Labrador; Maritimes)	Basking Shark (Atlantic)
		White Shark (Atlantic)	Acadian Redfish (Atlantic population)	Acadian Redfish (Bonne Bay population)
		Winter Skate (Southern Gulf of St. Lawrence)	Winter Skate (Eastern Scotian Shelf)	Darkblotched Rockfish (Pacific)
		Atlantic Bluefin Tuna	Quillback Rockfish (Pacific)	Yelloweye Rockfish (Pacific: inside waters; outside waters)
			Yellowmouth Rockfisk (Pacific)	Blue Shark (Atlantic)
			Canary Rockfish (Pacific)	Bluntnose Sixgill Shark (Pacific)
			Northern Wolffish (Atlantic)	Winter Skate (Western Scotian Shelf – Bay of Fundy)
			Spotted Wolffish (Atlantic)	Atlantic Wolffish
				Tope (Pacific)
				Longspine thornyhead (Pacific)
				Spotted Spiny Dogfish (Pacific)

Table 2 (*concluded*).

Extinct	Extirpated	Endangered	Threatened	Special concern
Diadromous Fishes				
	Striped Bass (St. Lawrence Estuary)	Atlantic Salmon (5 DUs)	Atlantic Salmon (South Newfoundland)	Atlantic Salmon (4 DUs)
		Coho Salmon (Interior Fraser)	Striped Bass (Southern Gulf of St. Lawrence; Bay of Fundy)	Bering Cisco (Arctic)
		Sockeye Salmon (Cultus Lake; Sakinaw Lake)	Chinook Salmon (Okanagan)	American Eel (Atlantic)
		Eulachon (Central Pacific; Fraser R.)	Eulachon (Nass and Skeena Rivers)	Green Sturgeon (Pacific)
		Atlantic Sturgeon (Maritimes; St. Lawrence)		Shortnose Sturgeon (Atlantic)
				Dolly Varden (Western Arctic)
Seabirds				
Great Auk (Atlantic)		Ivory Gull (Arctic)	Short-tailed Albatross (Pacific)	Black-footed Albatross (Pacific)
Labrador Duck		Roseate Tern (Atlantic)	Ross' Gull (Arctic)	Ancient Murrelet (Pacific)
			Marbled Murrelet (Pacific)	
			Pink-footed Shearwater (Pacific)	
Molluscs				
Eelgrass Limpet (Atlantic)		Northern Abalone (Pacific)	Atlantic Mud-piddock	Olympia Oyster (Pacific)
Reptiles				
		Leatherback Sea Turtle (Atlantic; Pacific)		
		Loggerhead Sea Turtle (Atlantic)		

Note: For each taxonomic species, the numbers of COSEWIC designatable units (DUs) is indicated in parentheses and separated by a semicolon.

coe-cde/cemam/report-rapport/sect4-eng.htm; accessed 10–4–2011), using $0.7N_0$ as a TRP and $0.3N_0$ as a LRP, where N_0 is the unfished abundance in numbers of individuals, rather than biomass, B_0.

5.4. Abundance trends

This section describes trends known or likely to have been driven by human and (or) environmental impacts, notably climate change and fishing. To detect trends in abundance, monitoring needs to be quite thorough, which is why our coverage of components and attributes of Canadian marine ecosystems is patchy. Full consideration of the observed and projected consequences to marine biodiversity driven by climate change and fisheries are addressed in sections 6 and 7, respectively.

5.4.1. Plankton

5.4.1.1. Pacific

Trend data are available for plankton at variable spatial and temporal scales in Canadian Pacific waters, although comparatively few pertain to phytoplankton. Based on data obtained in Barkley Sound, Vancouver Island, no trend in chlorophyll biomass is evident since data were first available in 2005 (Pawlowicz 2011). The longest time series of zooplankton data in Canadian Pacific waters extends from 1979 to the present and encompasses a transect extending from the BC continental margin to southern Vancouver Island (Fig. 42). Patterns in these southern BC waters, where zooplankton abundance and biomass are routinely estimated for more than 50 species, appear to be correlated with water temperature (Mackas et al. 2011). Relatively cool years (1980s, 1999–2002, 2007–2009) tend to favour boreal shelf copepods (small crustaceans, such as *Calanus marshallae*, *Pseudocalanus mimus*, *Acartia longiremis*) and Subarctic copepods (e.g., *Neocalanus plumchrus*, *N. cristatus*, *Eucalanus bungii*) and northern chaetognaths (transparent or translucent dart-shaped animals); relatively warm years (1983, 1993–1998, 2004–2005) tend to favour southern copepods (species whose ranges are centred 1000 km south of Vancouver Island) and southern chaetognaths. There is evidence to suggest that an abundant cool-water zooplankton community is associated with good local survival and growth of juvenile salmon and planktivorous seabirds (Mackas et al. 2007). Since 2000, there has also been an increase in the abundance of some gelatinous zooplankton, such as salps and doliolids (herbivorous planktonic tunicates) and medusae of jellyfish and ctenophores (which prey on other zooplankton and occasionally larval fish), and the warm-water planktonic snail *Clio pyramidata* (Fig. 43; Mackas et al. 2011).

In the Gulf of Alaska, mesozooplankton (which range in length from 200 μm to a few mm) have been sampled annually since 2000 as part of the Continuous Plankton Recorder (CPR) surveys. No significant annual trend in mesozooplankton abundance is evident in the past decade (although interannual variability can be considerable) (Batten 2011). There appear to be strong links between the species composition of mesozooplankton and temperature (which is related to the Pacific Decadal Oscillation (PDO; section 4). The proportional representation of small copepods tends to be higher when temperatures are relatively warm, whereas cold temperatures appear to favour large copepods (Batten 2011).

5.4.1.2. Atlantic

Temporal data on plankton are available for areas of the Canadian Atlantic. The longest time series of zooplankton data is available from the CPR surveys, which generally extend back to the early 1960s; these trends have been summarized by Head and Pepin (2009). Since 1990, phytoplankton abundance has generally been higher than the long-term average (1960–2006), although declines have been evident in recent years on the Scotian Shelf. The abundance of *Calanus finmarchicus* on the continental shelf is currently high on the Scotian Shelf and somewhat so on the Newfoundland Shelf, having returned to levels evident in the 1960s and 1970s following levels of low abundance in the 1990s. On the continental shelf, two Arctic species of *Calanus* (*C. glacialis*, *C. hyperboreus*) were more abundant in the 1990s and 2000s than in previous decades, although their abundance is generally low and highly variable among years. For smaller copepods (e.g., *Paracalanus* spp., *Pseudocalanaus* spp.), abundance has generally declined since the 1960s on the Scotian Shelf whereas an increase is evident on the Newfoundland Shelf. At present, euphausiids are generally below their long-term average abundance, which peaked in the 1970s.

The patterns revealed by the CPR data are evident on smaller spatial scales. For example, based on samples obtained from Station 27 (7 km east of St. John's harbour), chlorophyll concentrations (a metric of phytoplankton abundance) off Newfoundland in 2009 were at their highest levels since 2000, albeit returning to near-normal levels in 2010 (Pepin et al. 2011). A somewhat similar pattern is evident if one excludes the chlorophyll present during its period of peak abundance (i.e., during the spring "blooms"; Fig. 44). In the same region, and based on the abundance of eight dominant species of copepods, zooplankton appeared to be more abundant in 2009 and 2010 than the long-term average, although the 2008 estimate was the lowest in the time series, again reflecting high interannual variability in zooplankton biomass (Fig. 45). Specifically, the abundance of the small copepods (e.g., *Microcalanus* spp., *Oithona* spp., *Pseudocalanus* spp., *Oncaea* spp.) reached peak or near-peak levels while that of warm-water species such as *Acartia* spp., *Centropages* spp., and *Temora longicornis* were at low levels of abundance (Pepin et al. 2011). With the exception of *Calanus glacialis* and large copepod nauplii, most large copepods also increased significantly in 2009 and 2010. The temporal patterns of abundance observed at Station 27 are similar to those observed northeast of the Grand Banks (Flemish Cap) and on the southeast Grand Banks (Pepin et al. 2011). Interestingly, the duration of the spring phytoplankton bloom on the Scotian Shelf declined between 1999 and 2008 (based on data collected at the Halifax-2 fixed station, southeast of Halifax) concomitant with a general decline in "background" (outside the spring bloom) chlorophyll levels over the past 10 years (Fig. 46; Harrison et al. 2009).

5.4.2. Marine fishes

Trend data for marine fishes in Canadian waters can be obtained from two primary sources, both of which depend, to some extent, on DFO's fisheries-independent surveys. Population abundance or biomass data can be estimated from some form of sequential population analysis or statistical catch-at-age modelling (which incorporates catch-at-age data and as-

Fig. 42. Temporal trends in several groups of zooplankton species groups sampled from the BC continental margin to southern Vancouver Island. Data are presented as biomass anomalies, meaning that anomalies with positive values identify years with greater than the long-term average abundance. Circles indicate years with no or very few samples. The euphasiids (pelagic, shrimp-like crustaceans) in the lowermost panel are *Euphausia pacifica* and *Thysanoessa spinifera*. Source: Mackas et al. (2011).
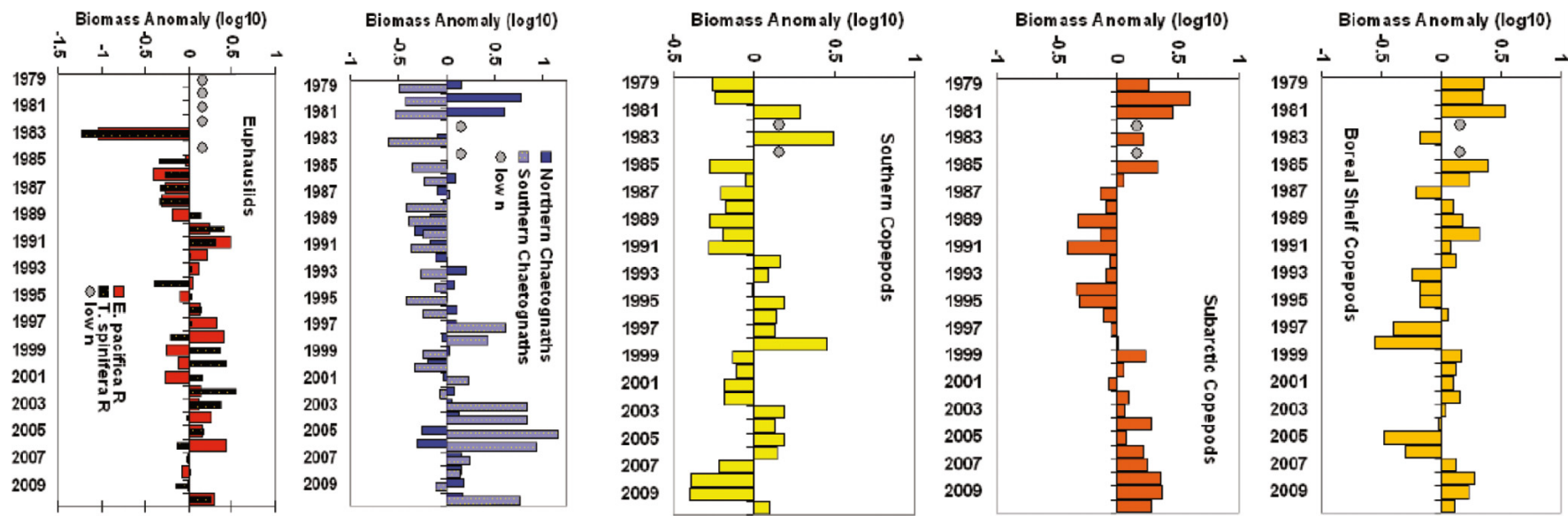
**Fig. 43.** Temporal trends in gelatinous zooplankton (doliolids, salps, jellyfish, and ctenophore medusae) and the warm-water planktonic snail (*Clio pyramidata*) sampled from the BC continental margin to southern Vancouver Island. Data are presented as biomass anomalies, meaning that anomalies with positive values identify years with greater than the long-term average abundance. Source: Mackas et al. (2011).
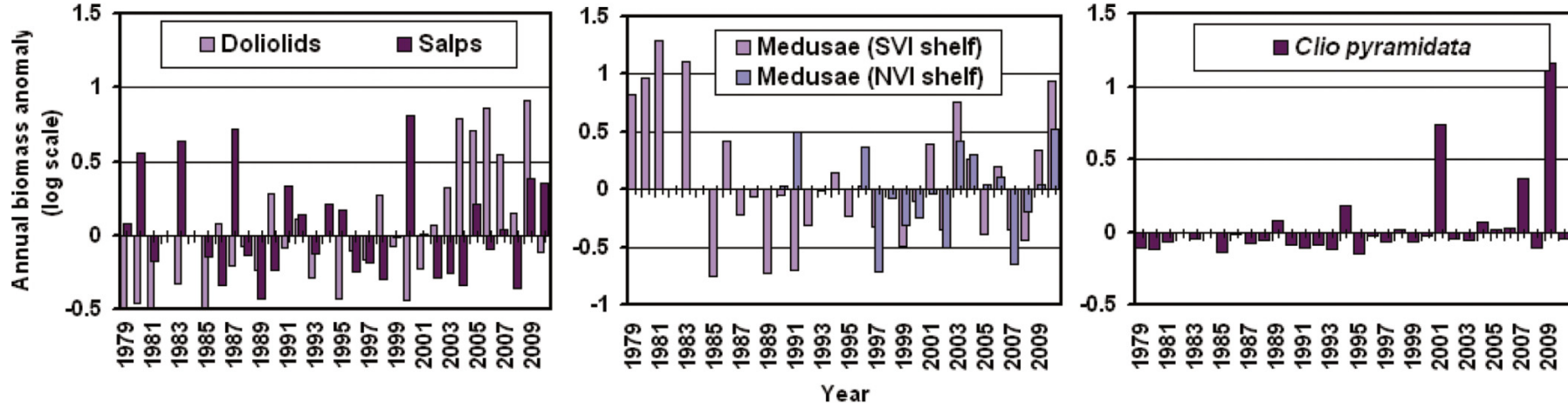
Fig. 44. Temporal trends in background chlorophyll a levels at Station 27 off St. John's, NL, during the time periods outside of the spring bloom periods (least squares annual averages ±1 standard error). Source: Pepin et al. (2011).
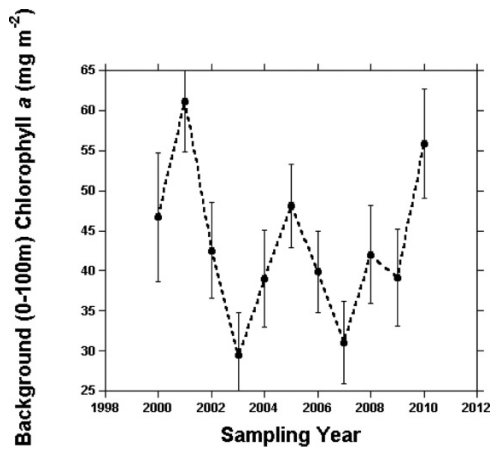
Fig. 45. Temporal trends in seasonally adjusted estimates of the mean biomass of eight dominant copepod species from Station 27 off St. John's (error bars represent one standard error). Source: Pepin et al. (2011).
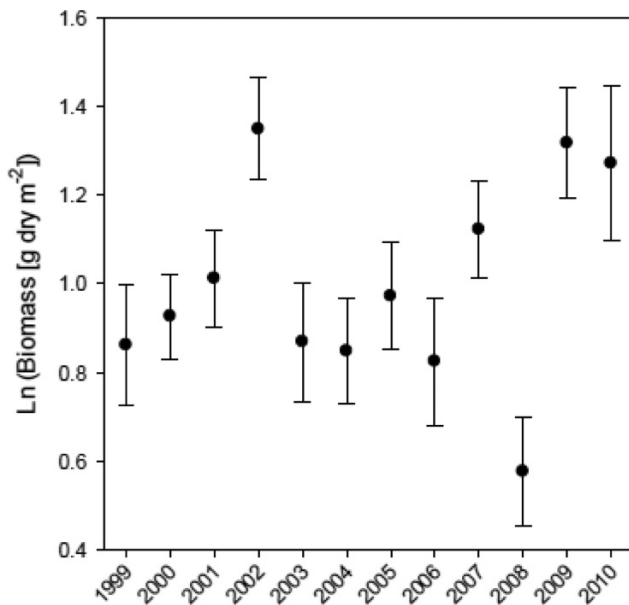
sumptions concerning fish natural mortality; the model output is often then "fitted" to the trend in survey catch rate), or they can be estimated directly from survey catch rates (such as numbers or weight of fish per unit of sampling effort, e.g., per tow of a bottom trawl net).

Restricting the analysis to marine fishes for which peer-reviewed stock assessments have been undertaken from 1970 to 2006, data are available for 40 populations of fishes that regularly inhabit Canadian waters (the full list of populations is given in Table 3). Although the distribution of most of these fishes lies entirely in Canadian waters, the distribution of others (and the spatial breadth of threats that

affect their sustainability) is considerably broader, e.g., Atlantic bluefin tuna, *Thunnus thynnus*. Multispecies abundance indices were constructed, following Hutchings et al. (2010). Although the baseline year used here maximizes the number of populations for which data are available, it should be noted that many species, particularly on the Atlantic coast, had already experienced fishing-induced reductions by 1970.

Overall, Canadian fish populations declined 52% between the first (1970–1974) and the last five years (2002–2006) in the time series (Fig. 47). The index was relatively stable from 1970 until the mid-1980s, declined considerably until the mid-1990s, and remained relatively stable thereafter. The trend differed between pelagic (mid-water) and demersal (bottom-dwelling) fishes. Following a period of increase through the 1970s and 1980s, pelagic fishes ($n = 16$) have declined to a level 40% lower than that of the early 1970s. By contrast, demersal fishes ($n = 24$) have shown a steady increase since the mid-1990s, although their levels are 58% lower than they were in the early 1970s.

The reductions evident throughout Canadian waters are also evident at smaller spatial scales. In the Canadian Pacific, demersal ($n = 13$) and pelagic fishes ($n = 6$) have declined 51% and 18%, respectively, since the 1970s. In the Canadian Atlantic, current levels of abundance indicate declines of 69% and 51% by demersal ($n = 11$) and pelagic ($n = 5$) species, respectively. The five High Seas migratory species that are caught in Canadian waters have declined 61% since the early 1970s.

Although Canada has not identified target reference points for most of its commercially exploited fishes, estimates of B_{MSY} for each stock, or population, can be made from surplus production models (Hilborn and Walters 1992). These estimates were reported by Worm et al. (2009) and Hutchings et al. (2010). In addition, for fishes whose distribution encompasses Canadian waters, but for which fisheries assessment responsibilities are the purview of the US or an international body, estimates of B_{MSY} are available from stock assessments. Multispecies abundance indices were constructed for the period extending from 1970 to 2007.

Restricting the multispecies indices to the 29 populations for which estimates of B/B_{MSY} are available for the time period under consideration, populations declined between 1970 and 2006 by an average 55% overall (from $B/B_{MSY} = 1.13$–0.51) and 42% and 61% for pelagic (0.80–0.47) and demersal (1.43–0.55) populations, respectively (Fig. 48). The decline in B/B_{MSY} ratios for Canadian Atlantic fishes is consistent with temporal patterns evident in the Northeast Atlantic, which are also estimated to be well below B_{MSY} (Hutchings et al. 2010). Similarly, demersal fishes in the Canadian Pacific are estimated to be above B_{MSY}, a pattern consistent with fishes throughout the northeast Pacific Ocean that are managed by the US (Hutchings et al. 2010). With the exception of Pacific demersal fishes, Canadian marine fishes are currently below the B_{MSY}. This includes the broadly distributed High Seas pelagic species that are regularly caught in Canadian waters.

At the species level, the decline of Atlantic cod is estimated to have been the greatest of any Canadian terrestrial or aquatic vertebrate (Hutchings and Rangeley 2011). The breeding population size is estimated to have declined by ~2 billion breeding individuals between the early 1960s and the mid-1990s, corresponding to a reduction in spawning stock biomass of roughly 2 million tonnes (Fig. 49). Notwithstanding recent,

Fig. 46. Dynamics of the spring phytoplankton bloom on the Scotian Shelf (based on data obtained from sampling station Halifax-2). (A) Timing, duration (based on 40 mg chlorophyll m^{-2} threshold for start and end of bloom), and magnitude of the bloom (as reflected by the numbers inside bars). (B) Background chlorophyll levels, i.e., outside of spring bloom periods (each error bar represents 1 standard error). Source: Harrison et al. (2009).
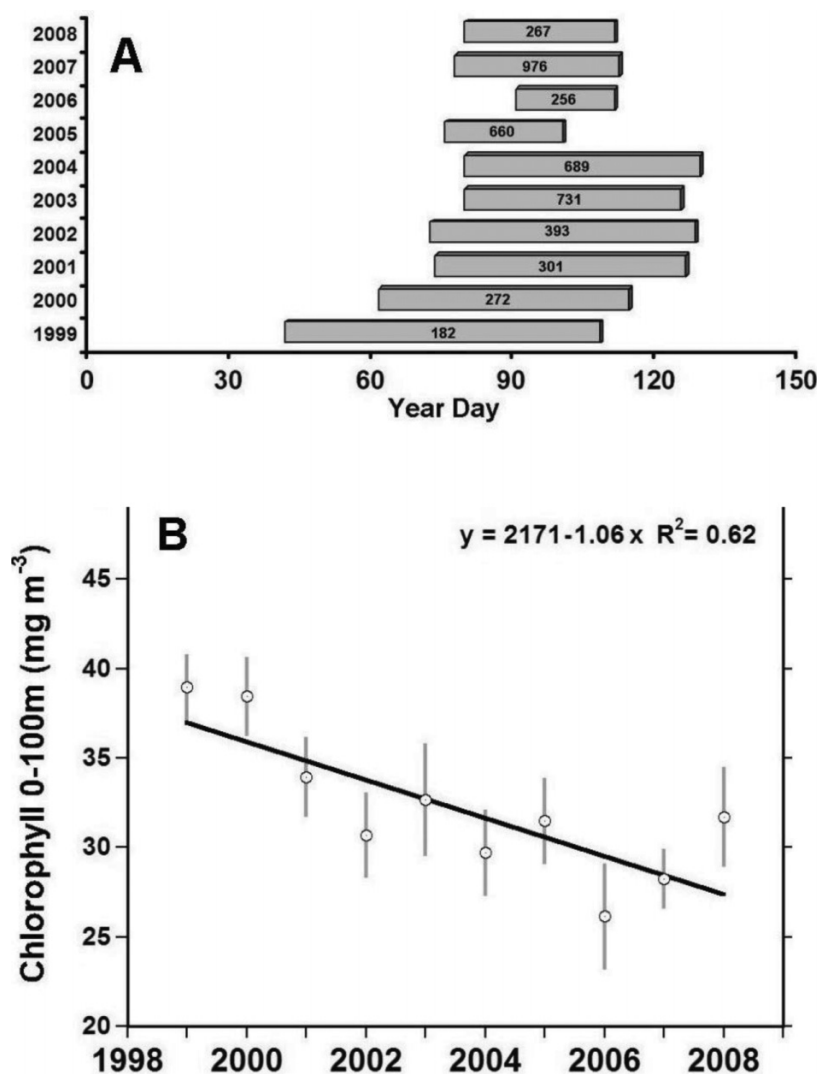
short-term positive trends in some areas (Frank et al. 2011; Hutchings and Rangeley 2011), meaningful recovery of the species has not been evident.

5.4.3. Diadromous fishes

5.4.3.1. Pacific

Within the marine and diadromous fishes in Canadian waters, Pacific salmon (*Oncorhynchus* spp.) are among the very few that can be termed "iconic" (others being Atlantic salmon and Atlantic cod in the east, Arctic char (*Salvelinus alpinus*) in the north). Five primary species have supported substantial fisheries for aboriginal, commercial, and recreational harvesters for more than a century: pink (*O. gorbuscha*), chum (*O. keta*), coho (*O. kisutch*), sockeye (*O. nerka*), and Chinook (*O. tshawytscha*). Pacific salmon are key species in the ecological dynamics of their freshwater and coastal marine habitats and are perceived by the general public as reflecting the quality of their freshwater and marine ecosystems.

The status of Pacific salmon depends on the spatial scale being considered. The last 50 years have been marked by a dramatic increase in the North Pacific of the total number of wild (i.e., nonhatchery) adults "returns" (i.e., catches plus spawners) of the three species for which the best long-term data are available: pink, chum, and sockeye (the North Pacific region includes populations from Korea, Japan, Russia, Alaska, Canada, and the conterminous US; Ruggerone et al. 2010a). On average, total annual abundances of wild pink and sockeye salmon populations increased by 60% and 56%, respectively, between the two decades 1952–1961 and 1996–2005, whereas total wild chum decreased by 20% (Fig. 50; Ruggerone et al. 2010a, 2010b). Canada's total wild populations of these species also tended to increase. Comparing the same two decades, pink, chum, and sockeye salmon in northern BC (the northern tip of Vancouver Island northward) have increased 78%, 16%, and 66%, respectively. This is considerably greater than two of those

Table 3. Marine fish stocks used in compiling: (*i*) the multispecies indices in Figs. 47 and 48; (*ii*) catch data in Fig. 65; and (*iii*) the multispecies indices of fishing mortality in Fig. 81.

| Species	Management unit	Years	Assessment body
Atlantic Ocean			
Atlantic Herring (*Clupea harengus*)	Scotian Shelf and Bay of Fundy	1965–2006	DFO
	4R Fall Spawners (Northern Gulf)	1971–2003	DFO
	4R Spring Spawners (Northern Gulf)	1963–2004	DFO
	4T Fall Spawners (Southern Gulf)	1974–2007	DFO
	4T Spring Spawners (Southern Gulf)	1974–2007	DFO
Atlantic Cod (*Gadus morhua*)	2J3KL (Northern Cod)	1962–1992	DFO
	3NO (Southern Grand Bank)	1953–2007	NAFO
	3Ps (St. Pierre Bank)	1959–2004	DFO
	3Pn4RS (Northern Gulf)	1964–2007	DFO
	4TVn (Southern Gulf, Sydney Bight)	1965–2009	DFO
	4VsW (Eastern Scotian Shelf)	1958–2002	DFO
	5Zjm (Georges Bank)	1978–2003	DFO
Haddock (*Melanogrammus aeglefinus*)	4X5Y (Bay of Fundy, Gulf of Maine)	1960–2003	DFO
	5Zejm (Georges Bank)	1968–2003	DFO
Pollock (*Pollachias virens*)	4VWX5Zc (Scotian Shelf to Georges Bank)	1974–2007	DFO
Cusk (*Brosme brosme*)	4X (Western Scotian Shelf, Bay of Fundy)	1970–2007	DFO
White Hake (*Urophycis tenuis*)	Scotian Shelf, Bay of Fundy, Georges Bank	1964–2005	Catch data only
American Plaice (*Hippoglossoides platessoides*)	23K (Newfoundland and Labrador)	1960–2004	DFO
	3LNO (Grand Bank)	1955–2007	NAFO
	3Ps (St. Pierre Bank)	1960–2005	Catch data only
Monkfish (*Lophius americanus*)	Labrador, Grand Bank, St. Pierre Bank	1977–2000	DFO
Redfish (*Sebastes* sp.)	23K (Newfoundland and Labrador)	1959–2001	Catch only
	3LN (Grand Bank)	1959–2008	NAFO
	3Pn4RSTVn (Gulf of St. Lawrence)	1953–2000	Catch data only
Greenland Halibut (*Reinhardtius hippoglossoides*)	01ABCDEF (Baffin Bay, Davis Strait)	1987–2006	Catch data only
	23KLMNO (Newfoundland and Labrador)	1975–2006	NAFO
	4RST (Gulf of St. Lawrence)	1970–2002	Catch data only
Yellowtail Flounder (*Limanda ferruginea*)	3LNO (Grand Bank)	1965–2009	NAFO
Albacore Tuna (*Thunnus alalunga*)	North Atlantic	1929–2005	ICCAT
Yellowfin Tuna (*T. albacares*)	Atlantic	1970–2006	ICCAT
Bigeye Tuna (*T. obesus*)	Atlantic	1950–2005	ICCAT
Bluefin Tuna (*T. thynnus*)	Western Atlantic	1969–2007	ICCAT
Swordfish (*Xiphias gladius*)	North Atlantic	1978–2007	ICCAT
Spiny Dogfish (*Squalus acanthias*)	Atlantic Coast	1990–2006	NMFS
Ocean Pout (*Zoarces americanus*)	Northeast Atlantic Coast		NMFS
Pacific Ocean			
Sablefish (*Aonoplopoma fimbria*)	Pacific Coast	1979–2004	DFO
Pacific Herring (*Clupea pallasii*)	Central Coast	1951–2007	DFO
	Prince Rupert District	1951–2007	DFO
	Queen Charlotte Islands	1951–2007	DFO
	Strait of Georgia	1951–2007	DFO
	West Coast of Vancouver Island	1951–2007	DFO
Pacific Cod (*Gadus macrocephalus*)	Hecate Strait	1956–2005	DFO
	West Coast of Vancouver Island	1945–2001	DFO
Rock Sole (*Lepidopsetta bilineata*)	Hecate Strait	1945–2001	DFO
English Sole (*Parophrys vetulus*)	Hecate Strait	1944–2001	DFO
Petrale Sole (*Eopsetta jordani*)	Northern Pacific Coast	1910–2005	NMFS
Pacific Hake (*Merluccius productus*)	Pacific Cast	1966–2008	NMFS
Longnose Skate (*Raja rhina*)	Pacific Coast	1915–2007	NMFS
Pacific Ocean Perch (*Sebastes alutus*)	Pacific Coast	1953–2007	NMFS
Darkblotched Rockfish (*S. crameri*)	Pacific Coast	1928–2007	NMFS
Widow Rockfish (*S. entomelas*)	Pacific Coast	1955–2006	NMFS
Yellowtail Rockfish (*S. flavidus*)	Pacific Coast	1967–2005	NMFS
Canary Rockfish (*S. pinnegar*)	Pacific coast	1916–2009	NMFS
Yelloweye Rockfish (*S. ruberrimus*)	Pacific Coast	1923–2006	NMFS

Table 3 (*concluded*).

Species	Management unit	Years	Assessment body
Shortspine Thornyhead (*Sebastolobus alascanus*)	Pacific Coast	1901–2005	NMFS
Longspine Thornyhead (*S. altivelis*)	Pacific Coast	1962–2005	NMFS
Lingcod (*Ophiodon elongates*)	Northern Pacific Coast	1956–2005	NMFS
Starry Flounder (*Platichthys stellatus*)	Northern Pacific Coast	1970–2005	NMFS
Pacific Sardine (*Sardinops sagax*)	Pacific Coast	1981–2005	NMFS
Pacific Chub Mackerel (*Scomber japonicas*)	Pacific Coast	1929–2008	NMFS

Note: Assessment body abbreviations: DFO, Department of Fsheries and Oceans; NAFO, Northwest Atlantic Fisheries Organization; ICCAT, International Commission for the Conservation of Atlantic Tunas; NMFS, US National Marine Fisheries Service.

Fig. 47. Trends in multispecies abundance indices for Canadian marine fishes, as reflected by changes in spawning stock biomass (SSB). A multispecies index for all populations and regions combined is shown in upper left panel. Remaining panels illustrate multispecies indices for pelagic (red) and demersal (green) populations separately. The solid lines represent the fixed-effect mean yearly estimates, based on a mixed-effects model with population as a random effect. The shaded regions represent the 95% confidence intervals on the fixed-effect mean. The number of stocks in each trend line is identified by "N". Full details of methods are available in Hutchings et al. (2010).
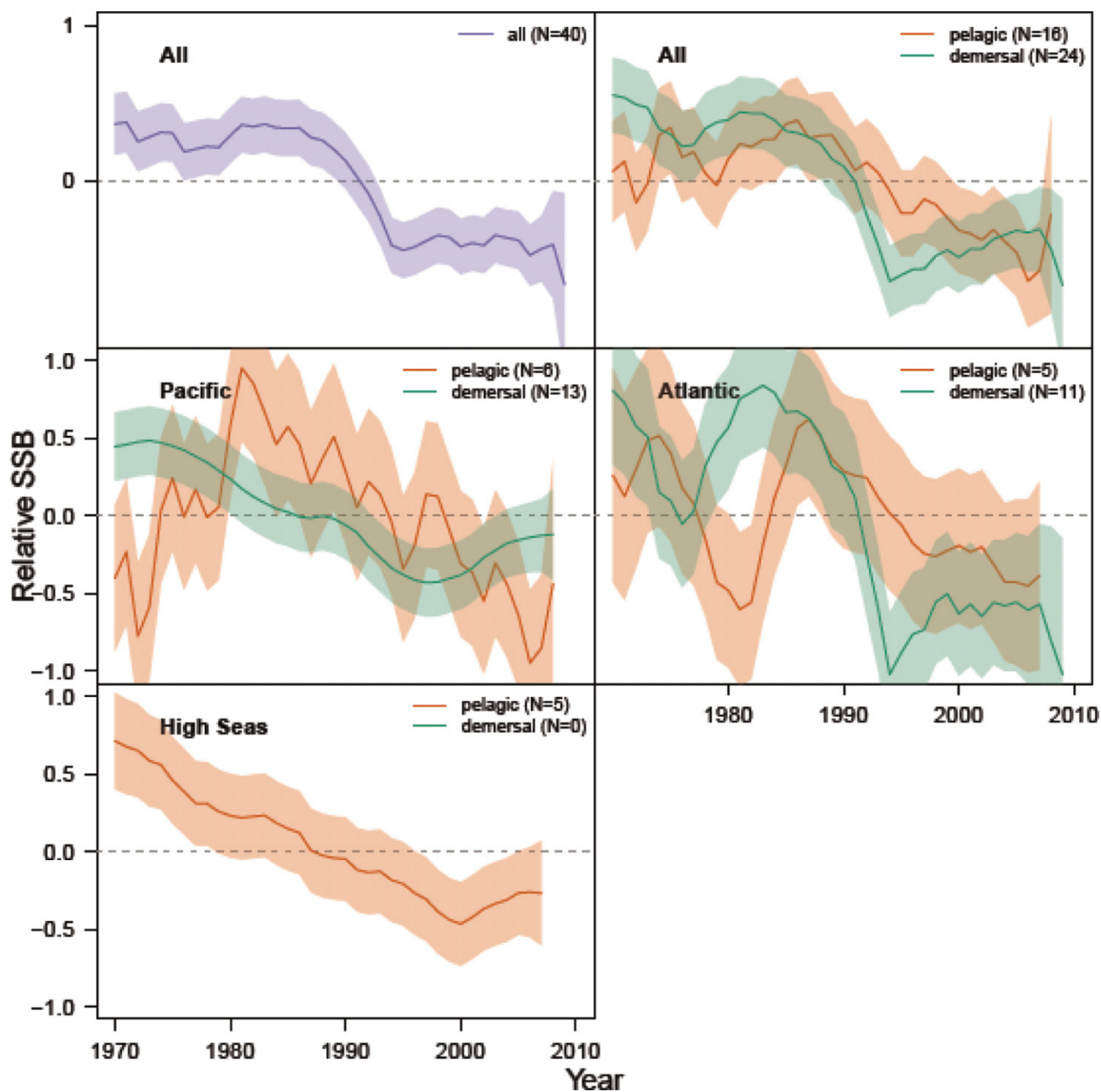
Fig. 48. Temporal trends in current biomass (B) relative to the estimated biomass at which the maximum sustainable yield (MSY) should be obtained (B_{MSY}). B_{MSY} is set to 1 in each panel (broken lines). Multispecies index for all populations and regions combined is shown in upper left panel. Remaining panels illustrate multispecies indices for pelagic (red) and demersal (green) populations separately. The solid lines represent the fixed-effect mean yearly estimates, based on a mixed-effects model with population as a random effect. The shaded regions represent the 95% confidence intervals on the fixed-effect mean. Full details of methods are available in Hutchings et al. (2010).
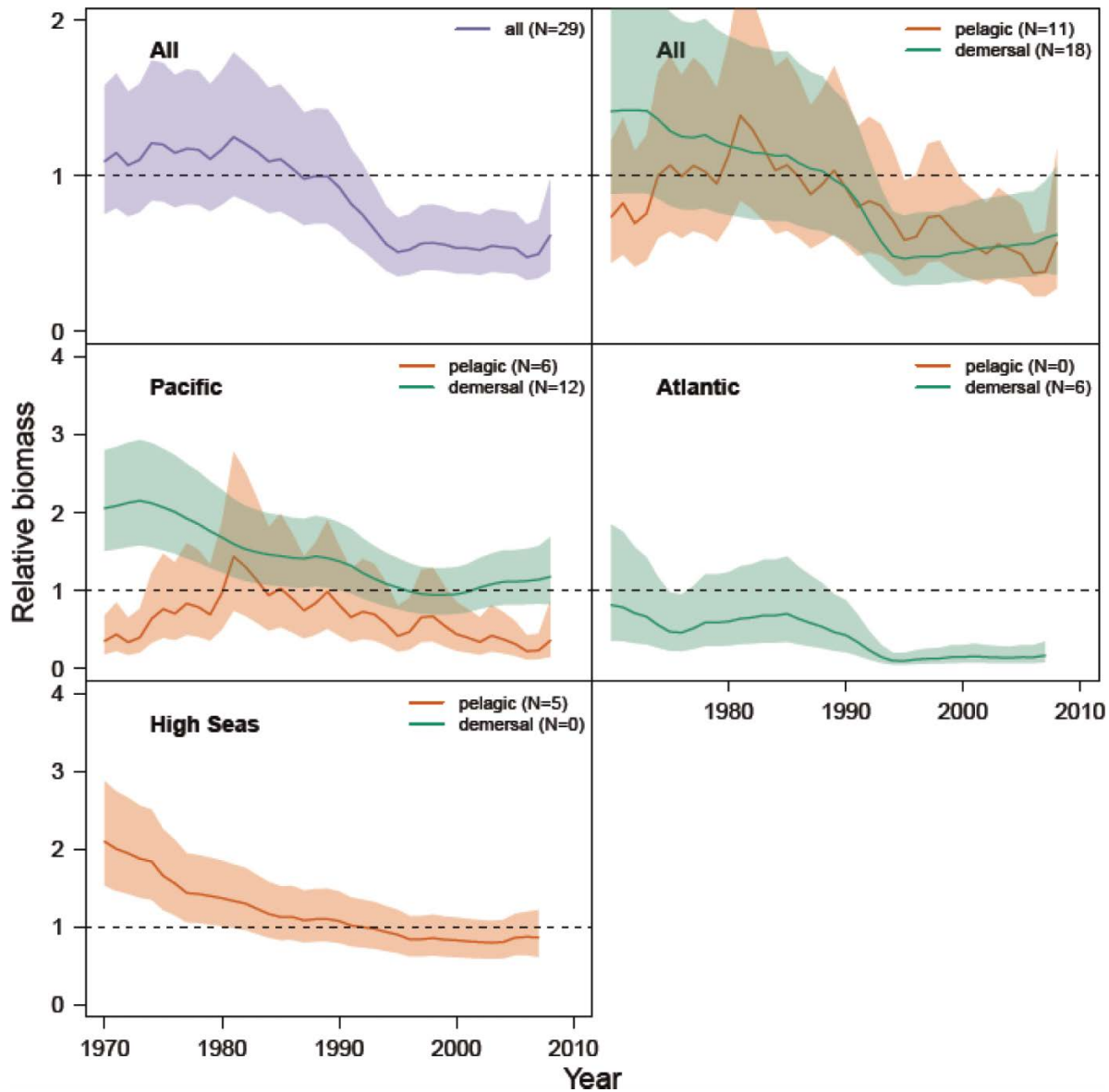
species in southern BC (19%, 61%, and 1%, respectively) (Fig. 51).

However, against this broad backdrop of general increases in regional aggregate abundance, there are numerous individual populations that show widely differing temporal trends, some exhibiting stability, some increasing, and others decreasing, sometimes considerably.

A key challenge for maintaining biological diversity among Pacific salmon stocks is that they are harvested in mixed-stock fisheries, in which adults from multiple populations of a given species return at the same time to fishing areas. These multiple populations are subjected to the same percentage harvest rate, yet some populations are more productive than others and can maintain relatively high spawner abundances despite high harvest rates. A good example of a mixed-stock fishery relates to the Cultus Lake sockeye population, which was assessed as endangered by COSEWIC in 2003 because of very low and decreasing abundance.

However, the Cultus stock was not listed for any protection under SARA because the Minister of Fisheries and Oceans concluded that a SARA listing would restrict commercial fisheries too severely on other, more abundant and productive populations that move through fishing zones at the same time as the Cultus stock. Despite the lack of SARA listing, DFO's Integrated Fisheries Management Plans for BC salmon contain restrictions on fisheries to take into account "stocks of con-

Fig. 49. Estimated spawning stock biomass of Canadian Atlantic cod from 1962 to 1998. Source: Hutchings and Rangeley 2011.
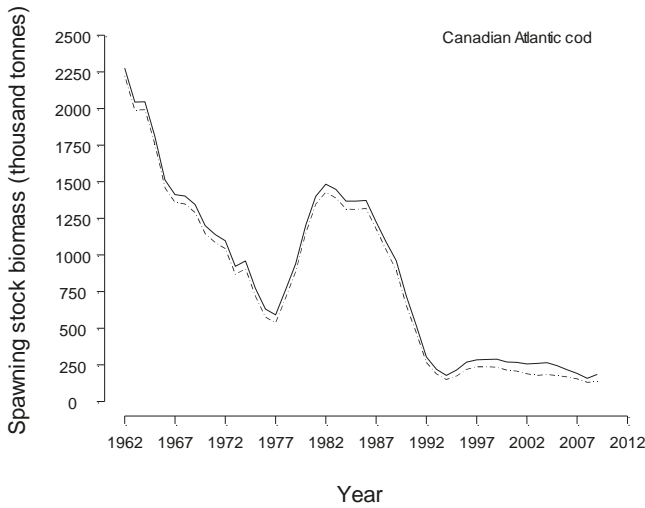
Fig. 50. Total annual abundance of adult returns (catch plus number of spawners, in millions) of wild pink, chum, and sockeye salmon for populations originating from the North Pacific Rim from 1952 to 2000. Note that vertical-axis scales differ on each graph.
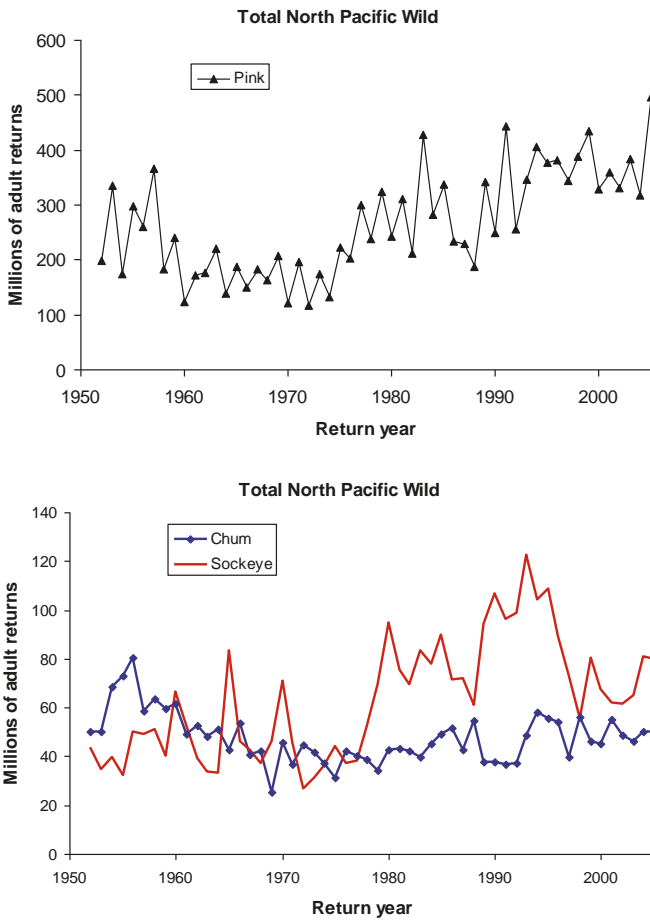
Fig. 51. Total annual abundance of adult returns (catch plus number of spawners, in millions) of wild pink, chum, and sockeye salmon for populations originating in northern and southern BC. Note that vertical-axis scales differ on each graph.
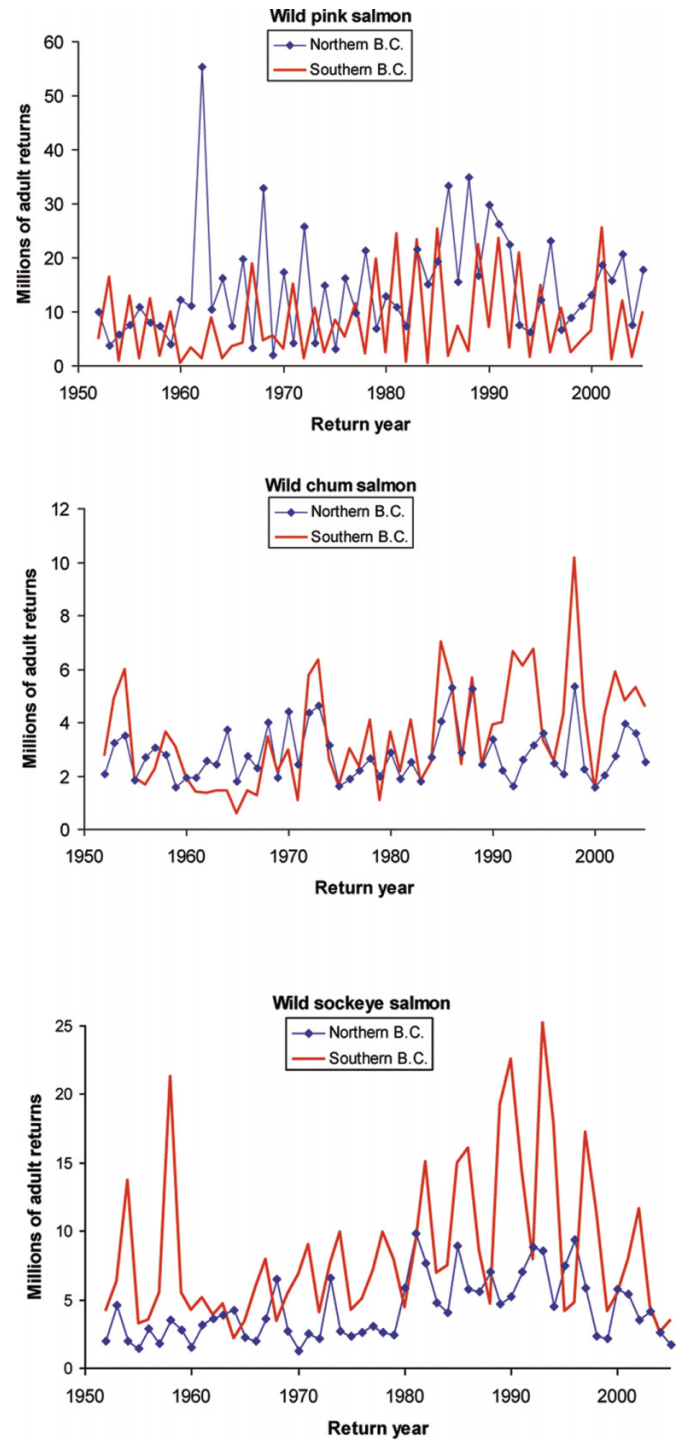

cern," including Cultus sockeye, where target harvest rates are about 30%, which is much lower than the 70%–80% rates in previous decades (DFO 2010b).

Large-scale salmon hatcheries exacerbate this mixed-stock fisheries problem by producing groups of fish that have higher productivity than their nearby wild counterparts and can with-

Fig. 52. Survival rate of wild and hatchery-origin juvenile coho salmon from southern BC populations during their residence in the ocean. "Brood year" is the year in which those fish were spawned.

Coho salmon, Southern British Columbia
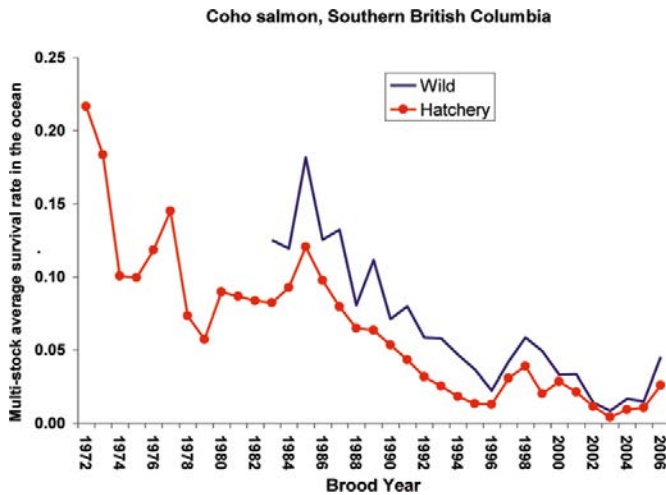
stand higher harvest rates. Although DFO has not constructed new large-scale "production" salmon hatcheries since 1985, the legacy of older hatcheries remains. For example, the percentage of hatchery-reared Strait of Georgia coho salmon that were caught in commercial and recreational fisheries increased from 0 to almost 80% in the 1975–2000 period (Sweeting et al. 2003). Also, it is not clear the extent to which these hatchery coho have augmented, as opposed to supplanted, wild coho salmon in the Strait. However, research on a similar problem concerning hatchery and wild pink salmon in Prince William Sound, Alaska, concluded that there was at least some replacement of wild stocks (e.g., Hilborn and Eggers 2001; Wertheimer et al. 2004).

Southern Interior BC coho salmon provide another example of Pacific salmon that are under stress. Due to declining rates of return associated with increasing human land use (Bradford and Irvine 2000) and decreasing survival in the ocean, commercial harvesting of these fishes has largely ceased since 1998. Exacerbating the problem of rebuilding these coho salmon populations are changing ocean conditions, which are reflected in southern BC waters by long-term decreases in survival of both wild and hatchery-origin juvenile coho salmon during their time at sea (Fig. 52).

The fifth major salmon species in BC is Chinook salmon, which has the largest body size of all Canadian salmon. Although population trends vary considerably, most major stocks in southern BC have decreased in abundance during the last decade (Figs. 53, 54).

An important challenge for meeting conservation goals is that it is logistically and financially impossible to monitor all populations of Pacific salmon. There are literally thousands of spawning sites and distinguishing fishes from all those individual sites in mixed-stock catches is not possible. Although stock identification methods exist through use of genetics, scale-growth patterns, or the presence of unique parasites, they are generally associated with the most commercially important stocks. Given this situation, both Canada and the US use numerous intensively monitored "indicator stocks" to indirectly represent the trends in survival rate and (or) abundance

of other nearby nonmonitored stocks of the same species. Although survival rate and (or) abundance levels tend to be, on average, positively correlated across such nearby populations, individual stocks can, and often do, move in different directions than indicator stocks. This can lead to drastic reductions, or even complete loss, of particular stocks without any remedial action being taken.

Overall, Pacific salmon in BC show a mix of time trends in abundance and survival rates, but there are serious and growing concerns about the conservation status of many sockeye, coho, and Chinook salmon stocks, especially in southern BC. These downward trends for many coho and sockeye stocks have continued, even after fishing was severely reduced or even eliminated. Causes of these downward trends are not clear, but a recent Expert Panel Report on the decline of Fraser River sockeye concluded that the causes most likely arise in the ocean, rather than in fresh water, and the primary candidates are marine food supply, marine predators, pathogens from fresh or marine waters, and competition with pink salmon in the high seas (Peterman et al. 2010). Another possible cause, termed delayed density dependence, appeared most relevant to only a subset of sockeye stocks (Peterman et al. 2010).

The Wild Salmon Policy (DFO 2005) outlines general objectives and strategies for maintaining a healthy and diverse set of wild salmon populations called conservation units (CUs). Work is still ongoing to identify appropriate "benchmarks" (analogous to reference points; see above) for the five Pacific salmon species. Those benchmarks will be used to categorize CUs into red (critical), amber (cautious), or green (healthy) states. Depending on available data, benchmarks will be set in terms of spawner abundance, trends in its abundance or occupancy of the spawning sites (Holt et al. 2009).

In addition to Pacific salmon, several other diadromous species inhabit coastal BC. Prominent among these are steelhead (*O. mykiss*), cutthroat trout (*O. clarki*), eulachon, green sturgeon (*Acipenser medirostris*), white sturgeon, and Dolly Varden (*Salvelinus malma*). Unfortunately, trend data are generally not available for BC's diadromous fishes, other than Pacific salmon. The exceptions tend to be those species assessed by COSEWIC. With a generation time of 30–40 years, the long-lived white sturgeon is estimated to have declined more than 50% in the past century and was assessed as endangered in 2003. Many populations of eulachon have experienced declines in excess of 90% in the past two decades, perhaps the most dramatic of these being the 98% reduction by the Fraser River population(s), resulting in COSEWIC status assessments in 2011 of threatened and endangered. The green sturgeon was assessed a status of special concern in 2004.

5.4.3.2. Atlantic

Diadromy is characteristic of the life histories of several native species of fishes in the Atlantic, including sea lamprey (*Petromyzon marinus*), two species of sturgeon (*Acipenser* spp.), American eel (*Anguilla rostrata*), several clupeids (e.g., shad), rainbow smelt (*Osmerus mordax*), Atlantic salmon, Arctic char, brook trout (*Salvelinus fontinalis*), and striped bass.

The wild species with the broadest spatial distribution in Atlantic Canada is Atlantic salmon, existing from Ungava Bay south to the American border. Between the early 1970s and the

Fig. 53. Total annual abundance of spawners ("escapement") and adult returns ("terminal run", which is the catch plus number of spawners in thousands) for BC Chinook salmon in the Nass River, Skeena River, and Harrison River fall Chinook, as well as the total Fraser River late terminal run. In the upper panel, "Above GW" refers to a specific location in the Nass River at Gitwinksihlkw. Note that vertical-axis scales differ on each graph.
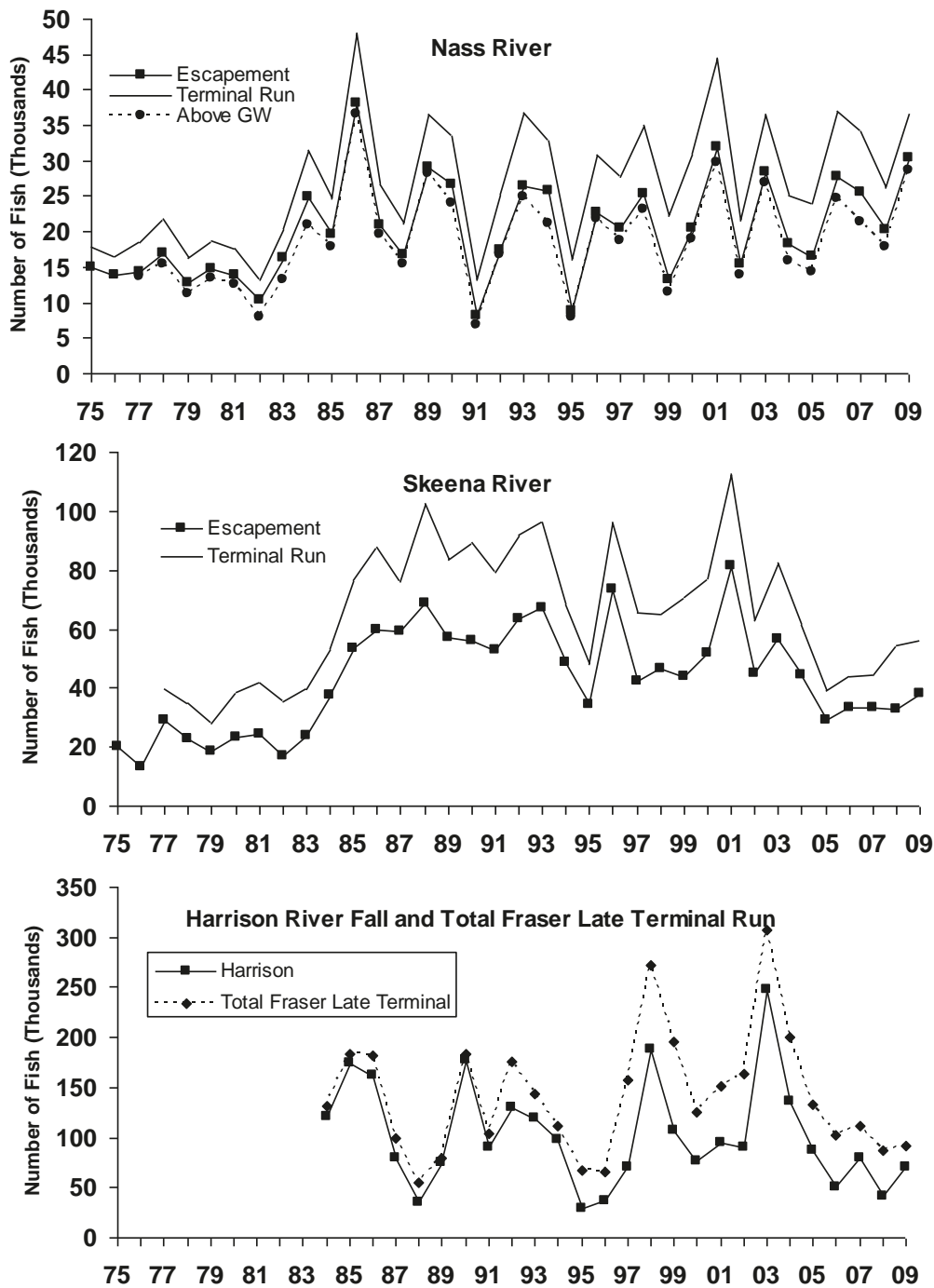
early 1990s, these salmon experienced a 70%–80% decline in the abundance of multi-sea-winter fish (i.e., salmon that spend more than one winter at sea prior to their return to their natal river) (Fig. 55); their abundance has remained stable since the mid-1990s, as has that of one-sea-winter salmon. It is important to note that these pan-Canadian trends in Atlantic salmon abundance mask significant spatial differences at smaller re-

gional scales. For example, many (albeit not all) river populations of salmon in Newfoundland and Labrador have increased since the closure of commercial fisheries in that area in 1992 (Fig. 56). Indeed, most were assessed as not at risk by COSEWIC in 2010. In contrast, many populations in Québec (Fig. 57), and most in the Maritimes (Fig. 58), have experienced significant declines. At the southern end of their Cana-

Fig. 54. Total annual abundance of spawners ("escapement") and adult returns ("terminal run", which is the catch plus number of spawners in thousands) for BC Chinook salmon in the Lower Strait of Georgia (upper panel). The lower panel shows abundance for two sets of index stocks for Chinook salmon, one based on 14 streams and another based on 6 streams that are used by the Pacific Salmon Commission (PSC) for the West Coast of Vancouver Island (WCVI).
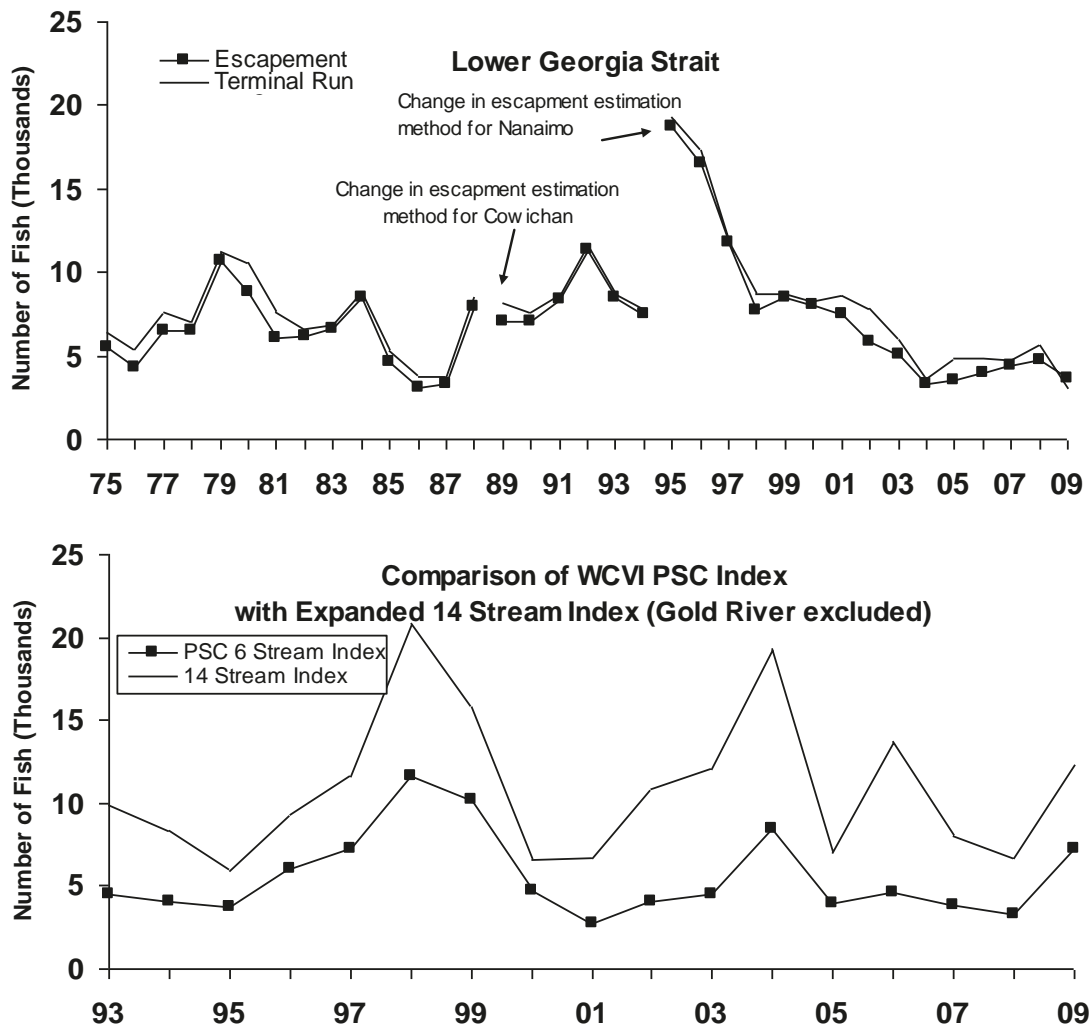
dian range, in the Bay of Fundy and along the southeastern coast of Nova Scotia, most Atlantic salmon populations have either been extirpated or are endangered.

Although temporal abundance estimates are generally not available for most other diadromous species in Atlantic Canada, existing data generally reveal declines. The numbers of eels returning to Lake Ontario via the St. Lawrence River have declined more than 95% since the 1970s (Fig. 59). Striped bass have been extirpated from the St. Lawrence Estuary (although recent introduction efforts may prove successful) and are deemed threatened in the southern Gulf of St. Lawrence and Bay of Fundy. As well, based on data available in COSEWIC species status reports (COSEWIC 2012a), sturgeon have generally declined during the past half-century. Since 2005, both Atlantic Canadian species have been assessed as being at risk: Atlantic sturgeon (*A. oxyrinchus*) is threatened (2011) and shortnose sturgeon (*A. brevirostrum*) is of special concern (2005).

5.4.4. Marine mammals

In contrast to temporal trends in marine and some diadromous fishes, many marine mammals exhibit evidence of dramatic increases following lengthy periods of exploitation (DFO 2010c). This is particularly true of pinnipeds, such as harp seals and grey seals (*Halichoerus grypus*) off the east coast (Fig. 60). Since the late 1960s, harp seals (which breed on ice flows from the Arctic south to the Gulf of St. Lawrence) increased from roughly 2 to 8 million in 2008 (DFO 2011a) and grey seals from ~10 000 in 1960 to as many as 430 000 today (DFO 2010d). Increases in abundance have also been evident among some cetaceans. The humpback whale (*Megaptera novaeangliae*), for example, on both Canada's east and west coasts has shown steady increases since the 1950s. Despite some impressive increases, other species have not fared as well. BC's killer whales were assessed as being at increased risk in recent years. Of the four DUs of killer whales recognised by COSEWIC, three have been assessed as

Fig. 55. Trends in the prefishery abundance of large (multi-sea-winter; black circles) and small (one-sea-winter; grey circles) Atlantic salmon to Canadian rivers. Source: COSEWIC (2011*a*).
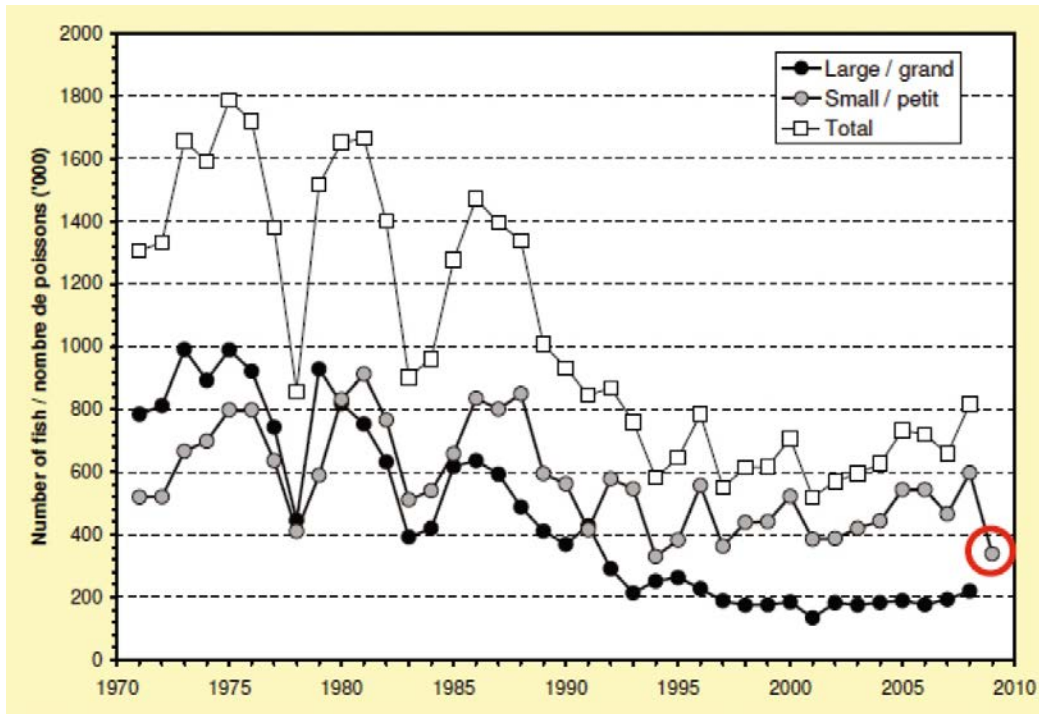
Fig. 56. Trends in the abundance of Atlantic salmon returning to selected rivers in Newfoundland and Labrador. Source: Gibson et al. (2006).
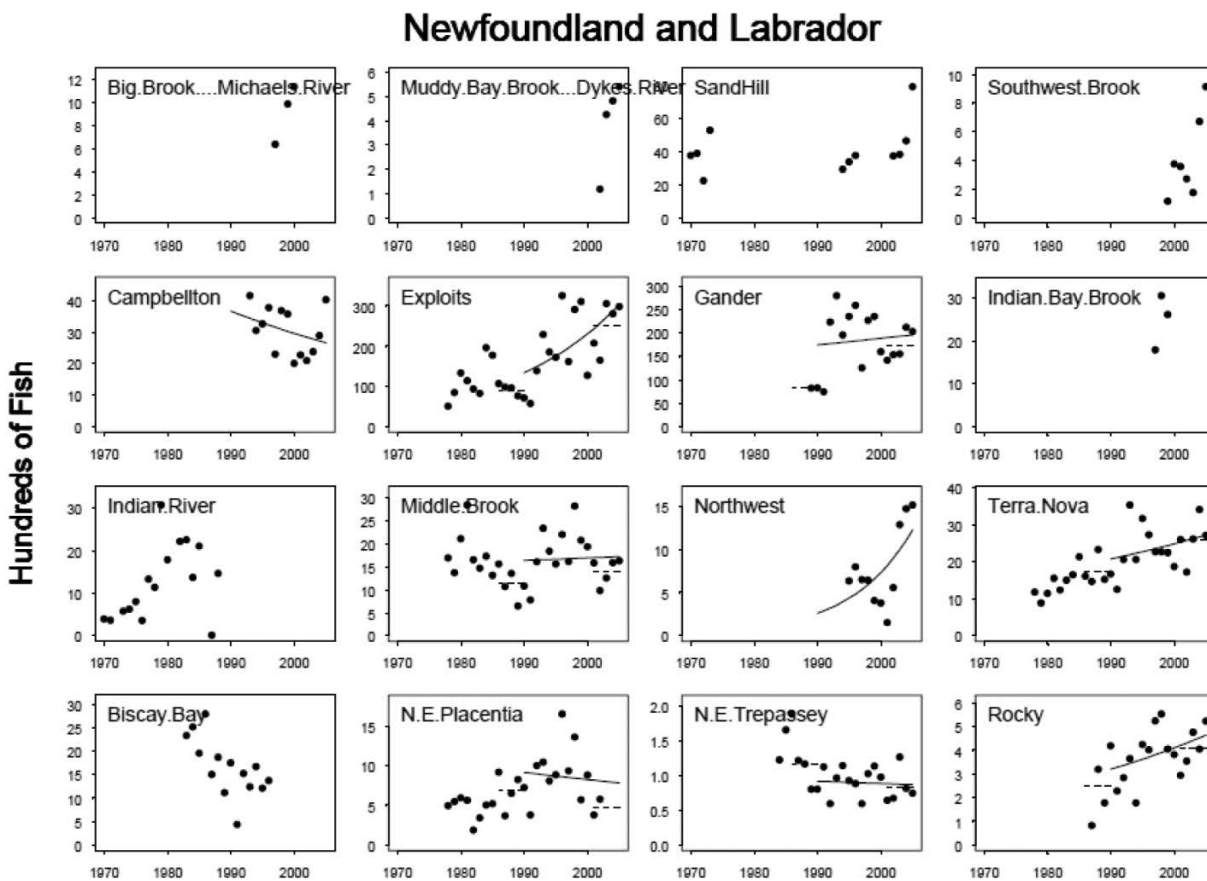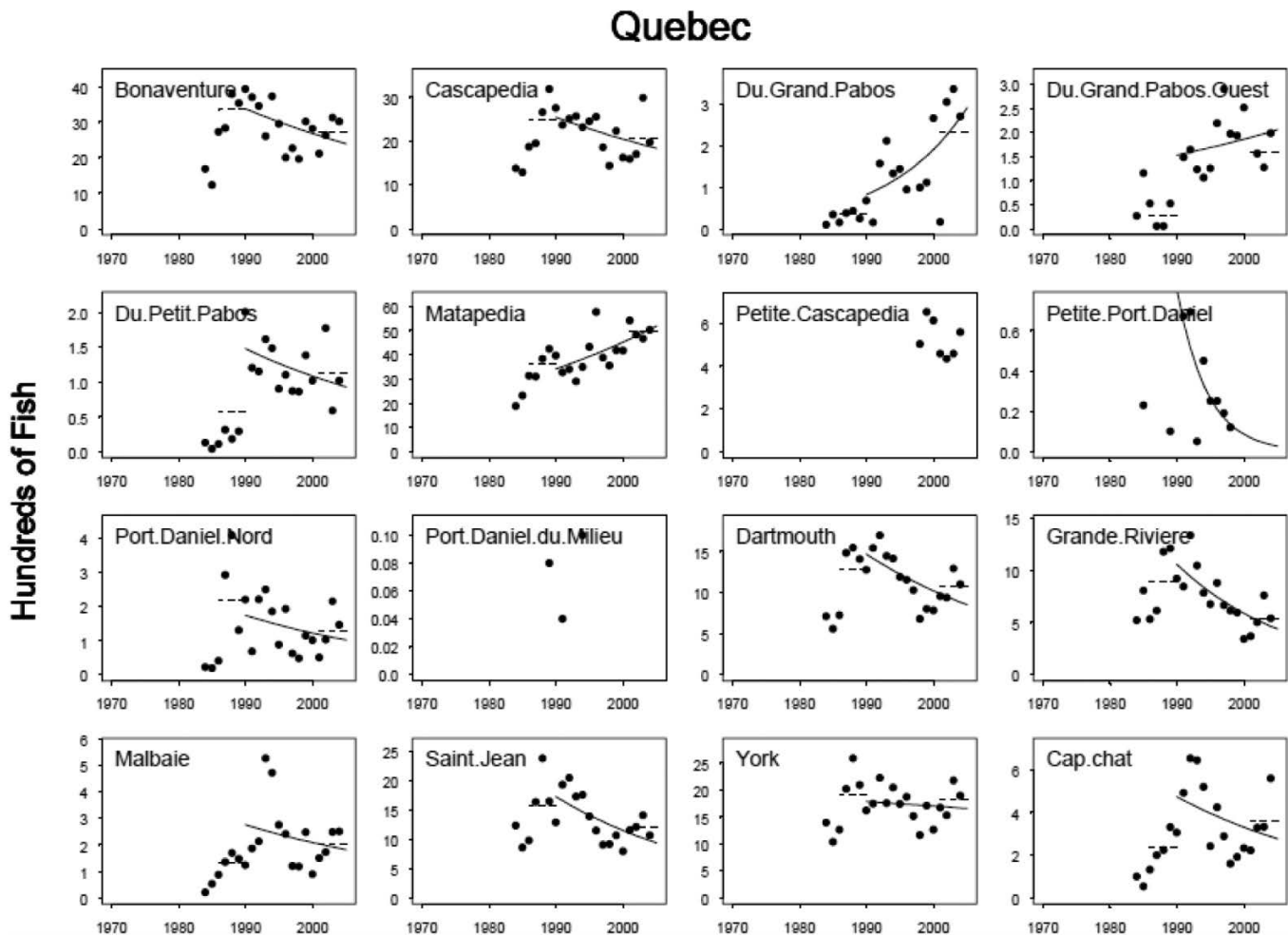
Fig. 57. Trends in the abundance of Atlantic salmon returning to selected rivers in Québec. Source: Gibson et al. (2006).

threatened (Northeast Pacific Offshore, Northern Resident, West Coast Transient) and one endangered (Southern Residents); these assessments are based primarily on small absolute numbers of reproductive maturely individuals. Populations of right whale (*Eubalaena glacialis*) and northern bottlenose whale (*Hyperoodon ampullatus*) in eastern Canada have both been assessed as endangered. As of January 2012, 35 wildlife species of marine mammals had been assessed as species at risk by COSEWIC.

5.4.5. Seabirds

Although the monitoring of some seabirds dates from the 1920s, abundance data were not systematically collected for many colonies until the 1980s (Gaston et al. 2009 provided an excellent overview of the population monitoring of Canadian seabirds). Trend data tend to originate from two sources. Ground-nesting birds, such as northern gannet (*Morus bassanus*), terns (*Sterna* spp.), and murres (*Uria* spp.), have been estimated either by counts made on the ground or from aerial photographs. Data on burrow nesters, such as storm petrels (Hydrobatidae) and various species of auks (Alcidae), have usually been obtained from transects and randomly placed sampling plots.

Gaston et al. (2009) recognized six zones for the purposes of estimating abundance of seabirds in Canadian waters: two in

each of the Pacific, Arctic, and Atlantic. On the Pacific coast, south of Queen Charlotte Sound, the most important breeding colony is that located on Triangle Island, where censuses have generally indicated declines from 1984 to 2004, based on abundance data for three burrow-nesting auks: rhinoceros auklet (*Cerorhinca monocerata*; 4% decline), Cassin's auklet (*Ptychoramphus aleuticus*; 46% decline), and tufted puffin (*Fratercula cirrhata*; 34% decline). In contrast to southern BC waters, burrow-nesting auks breeding north of Queen Charlotte Sound on Haida Gwaii have increased in abundance, e.g., ancient murrelets (*Synthliboramphus antiquus*; 66% from 1985 to 1995), rhinoceros auklets (34 and 90% from the mid-1980s to 2006 in two separate colonies) (Gaston et al. 2009).

According to Gaston et al. (2009), long-term data on seabirds in the Arctic are available for two primary areas: the central Arctic Archipelago (principally Prince Leopold Island in Barrow Strait and the Hell Gate – Cape Vera region between Devon and Ellesmere islands) and Digges and Coats islands at the mouth of Hudson Bay. At Prince Leopold Island, thick-billed murre (*Uria lomvia*) and black-legged kittiwake (*Rissa tridactyla*) have increased since the 1970s. Data of a more limited quality suggest that northern fulmars (*Fulmarus glacialis*) and glaucous gulls (*Larus hyperboreus*) might have

Fig. 58. Trends in the abundance of Atlantic salmon returning to selected rivers in the Maritimes. Source: Gibson et al. (2006).
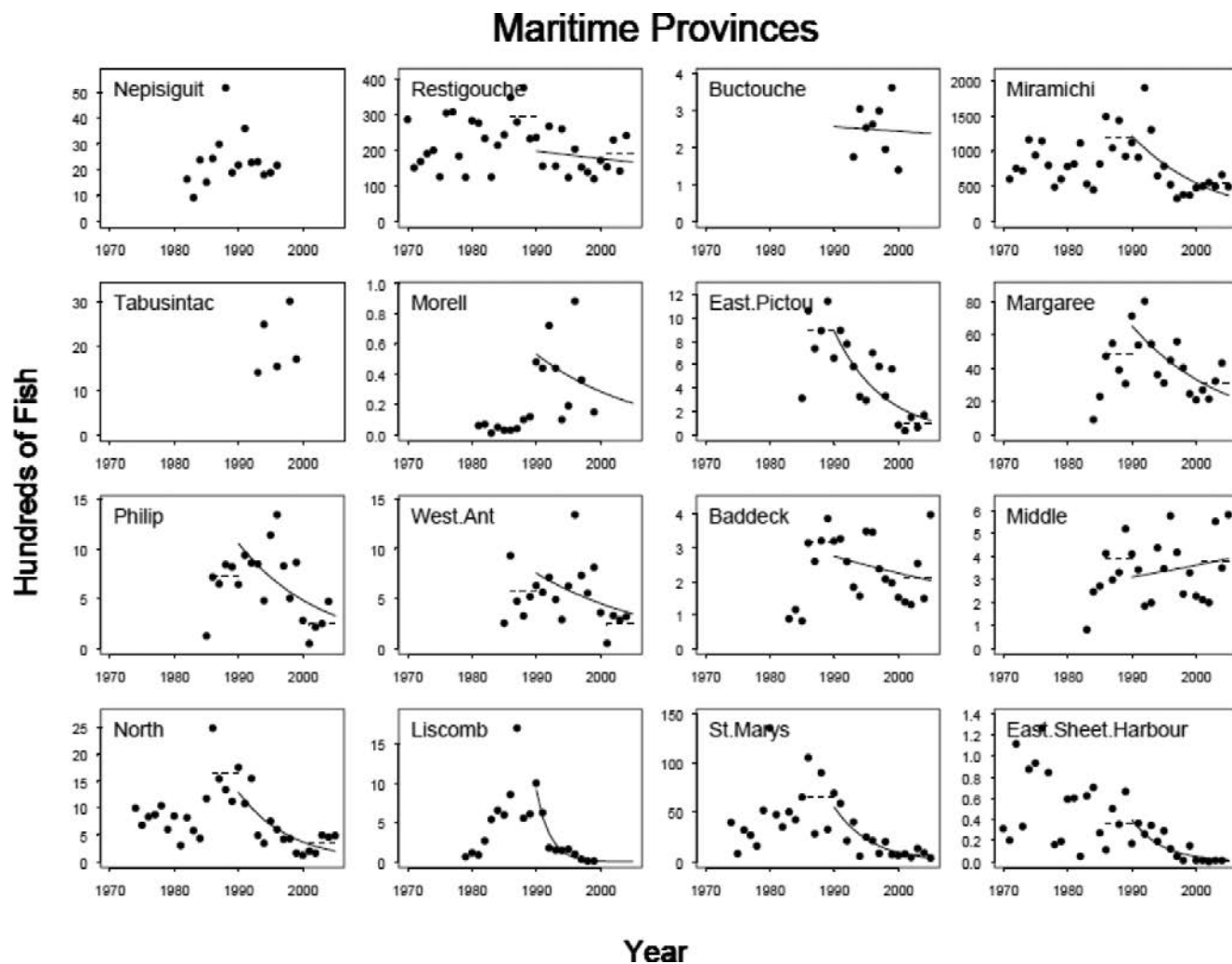
declined during this period. Perhaps the most dramatic population reduction has been experienced by ivory gulls (*Pagophila eburnea*) breeding in northern Nunavut. Formerly numbering 4000 breeding individuals, this population is estimated to have declined more than 80% between the 1980s and 2005 (Gilchrist and Mallory 2005), a rate sufficiently high to have resulted in the species being assessed as endangered by COSEWIC. Thick-billed murres have increased by approximately 33% on Coats and Digges islands between 1985 and 2007. By contrast, glaucous gulls and Arctic terns (*Sterna paradisaea*) may have declined more than 50% between the 1980s and 1997 in Hudson Bay.

In the Atlantic, northern gannet (the species for which abundance data are most extensive) have increased since the 1950s (Fig. 61). Most colonies of black-legged kittiwakes and common murres (*Uria aalge*) appear to have increased from 1970 to the early 1990s and declined moderately thereafter.

As of January 2012, COSEWIC had assessed eight seabirds as being at risk. However, only five of these breed in Canada (ivory gull, roseate tern (*Sterna dougallii*), marbled murrelet, ancient murrelet, Ross' gull (*Rhodostethia rosea*)).

6. Trends in Canadian marine fisheries and aquaculture

6.1. Introduction

A key objective of this section is to lay groundwork for some of the sections that follow. That is, before one can address the projected consequences to marine biodiversity of fisheries (section 8) and aquaculture (section 9), one requires descriptions of the past and present magnitude of these potential stressors. To provide perspective, the section begins by briefly examining global trends in capture fisheries and aquaculture before turning to the trends in Canada, including a consideration of their respective monetary value.

As of 2009, it is estimated that capture fisheries and aquaculture supplied the world with 145 million tonnes of fish and aquatic invertebrates (Fig. 62), with Canada supplying ~1% of this amount. At present, Canada ranks 20th among nations in fisheries catches. However, in terms of monetary value, Canada ranks 7th globally, with the vast majority of its production used for human consumption. Approximately 117.8 million tonnes of the global catch in 2009 was for human consump-

Fig. 59. Estimated number of American eels ascending the Moses–Saunders eel ladder in the Upper St. Lawrence River at Cornwall, ON, and New York Power Authority (Massena, NY) eel ladder (data are combined for both ladders). Source: COSEWIC (2012b).
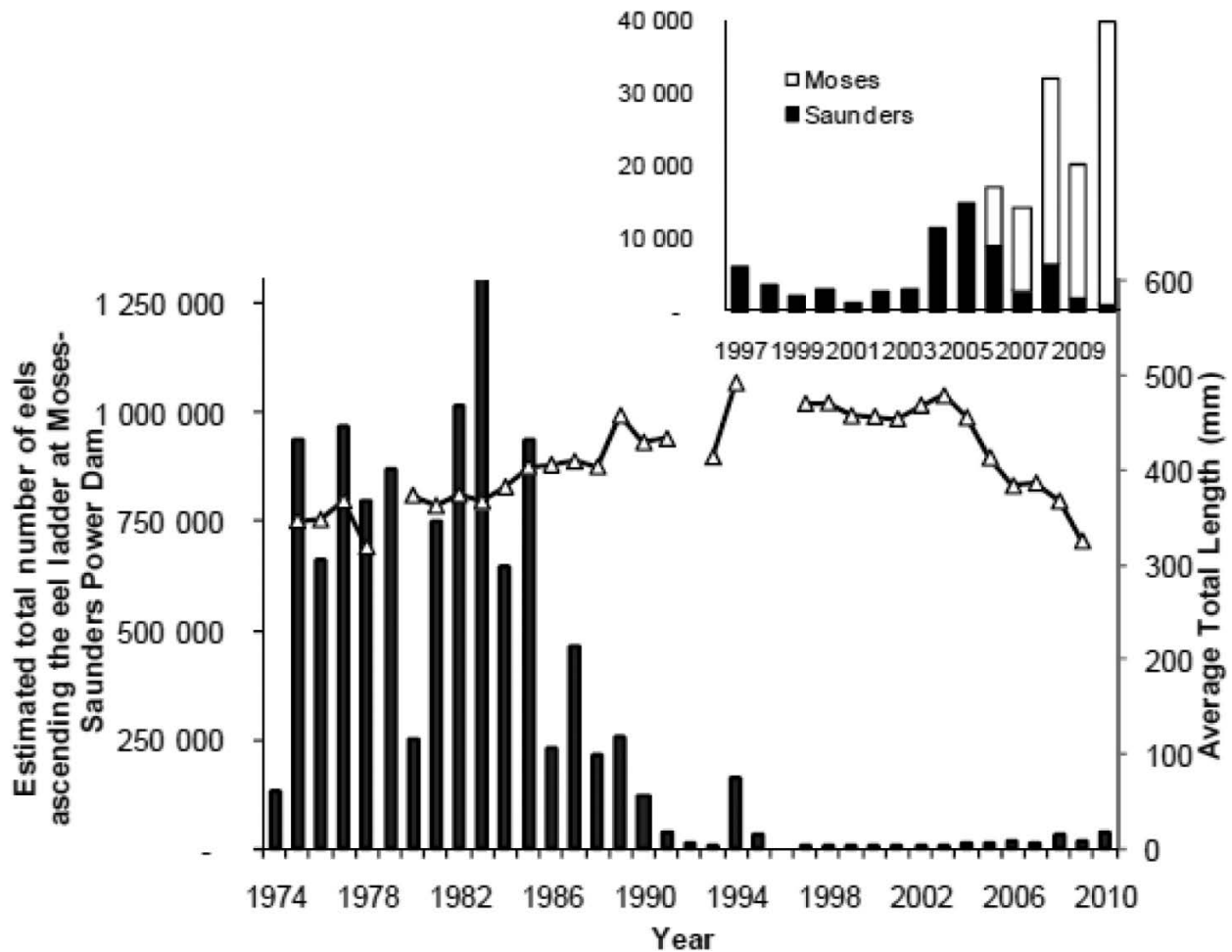
tion, resulting in an average food supply of 17.2 kg of fish and aquatic invertebrates per person (FAO 2010).

Capture fisheries supplied about 90 million tonnes of the global fish and aquatic invertebrate production in 2009, 89% of which derived from marine fisheries (Fig. 62). Marine capture fisheries production has been relatively stable over the past decade. Increases in global fisheries production during that period have come from aquaculture, both marine and inland. In fact, aquaculture is the fastest growing animal-food-producing sector globally and, given the 68 million tonnes produced in 2008, it could overtake capture fisheries as the main source of food derived from fish (and invertebrates) as early as 2012, assuming that its annual rate of growth of 6.6% in 2008 (FAO 2010) is maintained and capture fishery landings remain stable. Approximately 47% of the fish and aquatic invertebrates for human consumption in 2009 were derived from aquaculture (FAO 2010), with 35.0 million tonnes supplied by inland aquaculture and 20.1 million tonnes supplied by marine aquaculture (Fig. 62). Similarly, in Canada, catches from capture fisheries have remained relatively stable or have declined somewhat over the past decades (Fig. 63), whereas aquaculture production has increased (Fig. 64). Aquaculture in Canada, however, still only contributes about 15% of the country's total fisheries production.

6.2. Capture fisheries

6.2.1. Landed catches in Canada

Data on fisheries landings by Canadian enterprises are generally available for catches made in Canadian waters since the mid-1970s. Detailed data on tonnage caught on a species-by-species basis are available for 1990–2009. (Data of an instructive albeit more limited nature are available for earlier years, e.g., Fig. 65) These detailed data allow for catches to be apportioned to demersal (bottom-dwelling) fishes, pelagic (mid-water) fishes, diadromous fishes (e.g., Pacific salmon, alewife (*Alosa pseudoherangus*)), invertebrates (dominated by lobster (*Homarus americanus*) and snow crab (*Chionoecetes opilio*)), and other species (e.g., plants, sea cucumbers (Holothuriidae), sea urchins (e.g., *Strongylocentrotus droebachiensis*)). When evaluating the magnitudes of these catches, it is important to note that these landings data do not account for fish that were illegally captured or discarded.

In 2009, Canadian fishing enterprises landed 924 757 tonnes, the 3rd lowest catch in Canadian waters since 1976 and almost certainly in the past half-century. The only years since 1976 in which catches were lower were 1995 (857 310 tonnes) and 1996 (918 662 tonnes). Since Canada extended its fisheries jurisdiction to 200 nautical miles in 1977, the trends

Fig. 60. Abundance trends for selected species of marine mammals. Source: DFO (2010c).
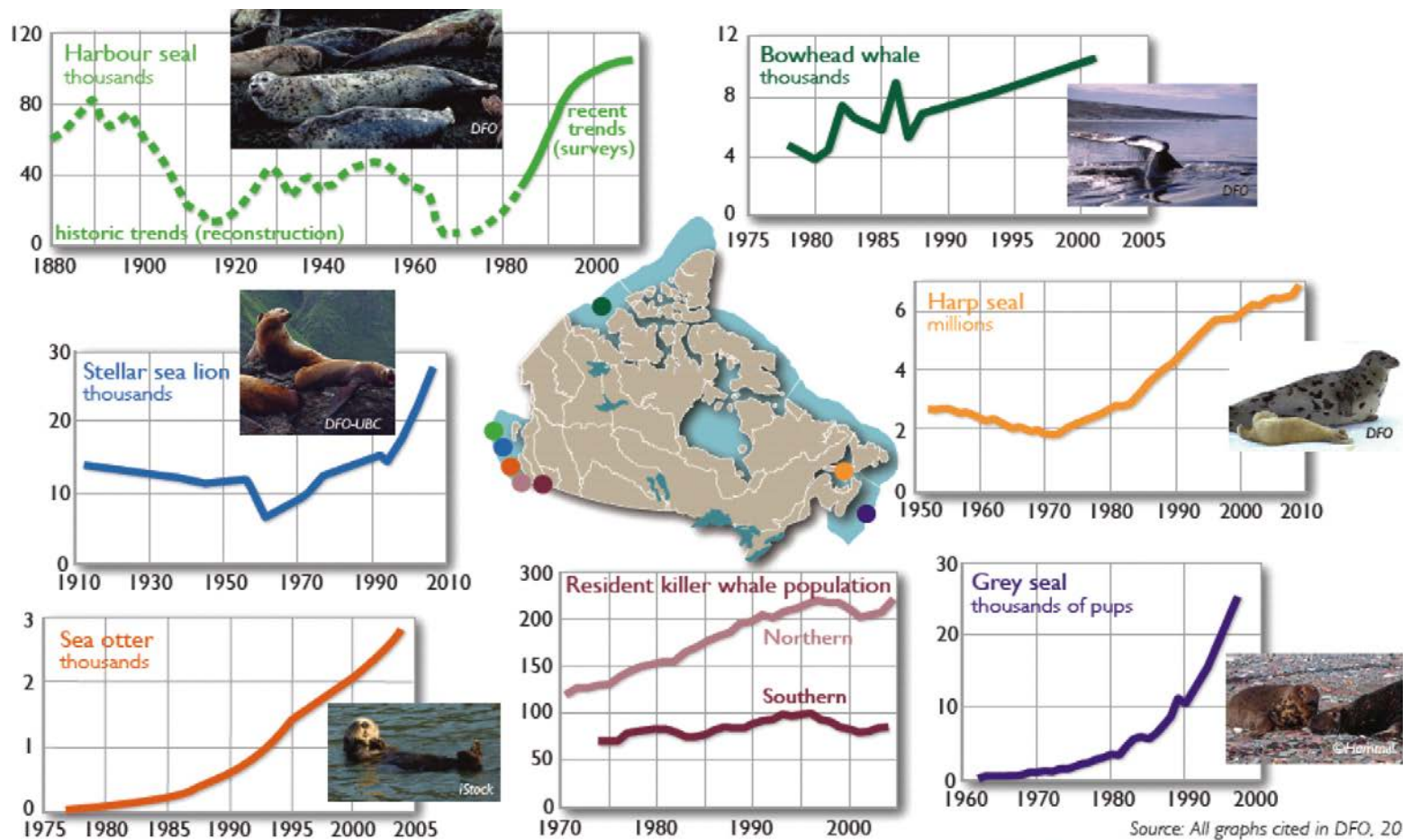
Fig. 61. Trends in northern gannet abundance at colonies in the Gulf of St. Lawrence and off eastern Newfoundland. Source: Gaston et al. (2009).
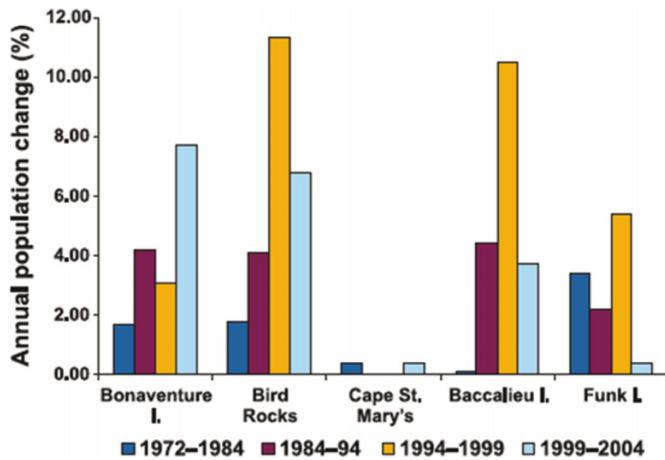
Fig. 62. Contribution of marine capture fisheries and marine aquaculture to world fisheries production during the last two decades. Data source: FAO (2010).

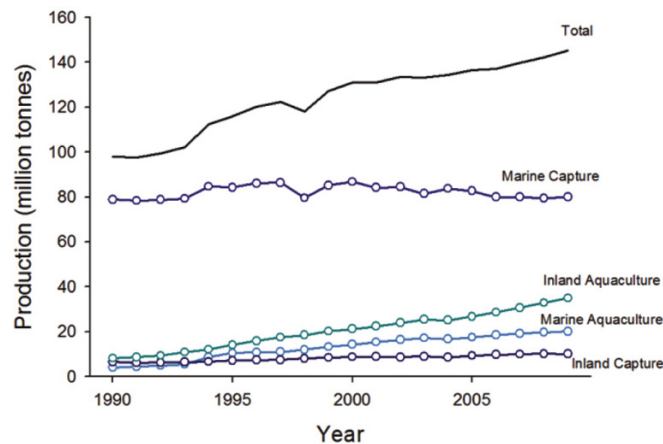in catches from all sources have differed between Atlantic and Pacific fisheries. After reaching a peak in 1988 (1 385 137 tonnes), Atlantic catches declined by approximately half; the total Atlantic catch in 2009 was 767 573 tonnes (Fig. 63). Between 1976 and 2009, catches in Pacific waters fluctuated about a mean of 217 324 tonnes. Following a near doubling during the 1980s, catches in Pacific waters declined from a peak of 312 104 tonnes in 1991 to 157 184 tonnes in 2009.

Canada's fisheries differ considerably in terms of species composition. The Atlantic fishery, once dominated by demersal fishes such as cod (note the catches of demersal fishes prior to 1990; Fig. 65), is now predominantly comprised of invertebrates. Since the 1994 minimum for Atlantic Canada, the annual tonnage of invertebrate catches increased to roughly 400 000 tonnes, while those for the pelagic and demersal fisheries remained comparatively stable at 300 000 tonnes and 200 000 tonnes, respectively. From 1990 to 2009, the landed catches of the Pacific fishery were dominated by demersal

fishes, averaging 137 984 tonnes annually. Catches of Pacific diadromous fishes (being composed entirely of Pacific salmon) declined markedly from almost 100 000 tonnes in 1990 to a low of 5 373 tonnes in 2009. In contrast to the Canadian Atlantic, Pacific landings of pelagic fishes and invertebrates have been comparatively low and have not experienced systematic trends over time.

6.2.2. Value of landed catches in Canada

The landed value of Canada's commercial fisheries in 2009 was among the lowest values since Canada's responsibility for fisheries management was extended from 12 to 200 nautical miles in 1977 (Fig. 66). Corrected for inflation and reported in 2011 dollars, Canadian landings from the Atlantic and Pacific coasts combined were valued at $1.66 billion in 1977 and $1.72 billion in 2009. At its height, the landed value of Canadian fisheries was $2.78 billion in 1987 and $2.69 billion in 2000. Concomitant with these changes is an increase in the export of wild fisheries products, from ~30% of production in 1990 to ~46% in 2009. The value of the Atlantic fisheries has always exceeded that of the Pacific fisheries (Fig. 66). The latter has been declining since the late 1970s, primarily because of a reduction in salmon. In contrast, the Atlantic fisheries increased steadily in value until the early 2000s (because of the increased value of the invertebrate fisheries), after which it declined.
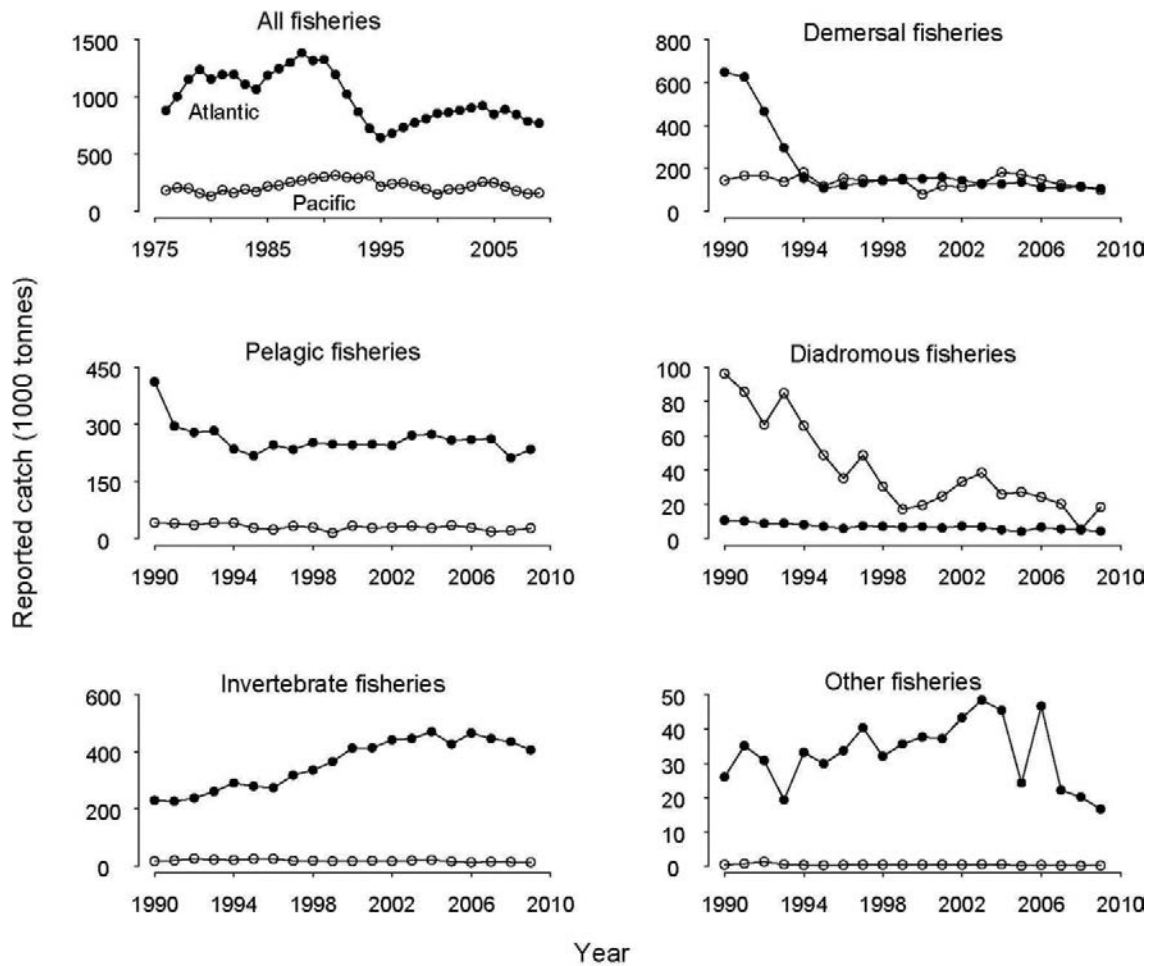
6.3. Aquaculture

6.3.1. Marine aquaculture production in Canada

Marine aquaculture in Canada began to develop in earnest in the 1980s. Initially dominated by shellfish, it was rapidly surpassed by finfish production (Fig. 64). (The word "shellfish" is now widely used in the fisheries and aquaculture literature to refer to invertebrates, such as blue mussels, *Mytilus edulis*, and lobster, although none of these "shellfish" is actually a fish, which are often identified as "finfish.") The expansion of aquaculture, driven largely by finfish production, was rapid until 2002 when production then stabilized (Fig. 64), a "stalled expansion" attributed to multiple factors, including disease, government-imposed moratoria in response to environmental concerns, and market considerations. Industry believes that opportunity for expansion remains and production could more than triple by 2015 (CAIA 2012).

Marine finfish aquaculture in Canada has been dominated by the production of salmonid fishes (Atlantic salmon, Chinook salmon, coho salmon and steelhead trout), although more recently small-scale aquaculture of other marine species (Atlantic halibut, Atlantic cod, sablefish (*Anopoploma fimbria*), spotted wolfish (*Anarhichas minor*), and Atlantic wolfish (*A. lupus*)) has been initiated. The major finfish aquaculture region is BC, where more than two thirds of the country's production is located (Fig. 67). Finfish aquaculture on the Pacific coast has developed rapidly since the mid-1980s and now involves about 130 licensed tenures encompassing a total area of 4575 ha. Approximately 100 of the tenures are currently active, with typically 80% of these tenures operational in any one year. New Brunswick remains Canada's 2nd largest finfish producer, though recent rapid expansion of operations in Newfoundland may soon challenge this ranking. Newfoundland has more than doubled its finfish aquaculture production in the last 5 years from 5006 tonnes in 2005 to 12 899 tonnes in 2010 (DFA 2010).

Fig. 63. Reported catches from various fisheries in the Atlantic (●) and Pacific (○) regions. Data source: DFO (2012).

Fig. 64. Marine aquaculture production in Canada. Data source: DFO (2012).
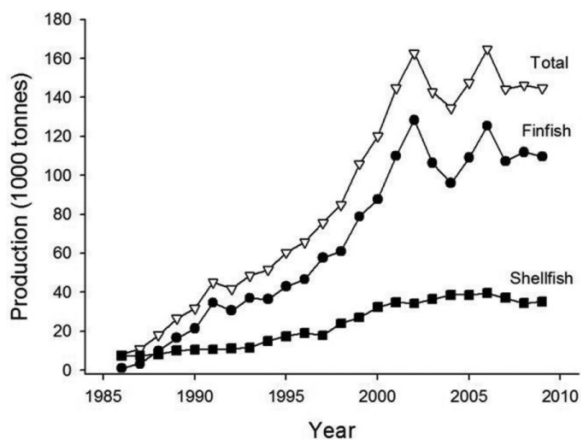
Fig. 65. Landings (tonnes) reported from Canadian Atlantic fisheries for demersal (bottom-dwelling) fishes from 1970 to 2004. Data are from DFO's stock assessments for Atlantic Canadian demersal fishes listed in Table 3.
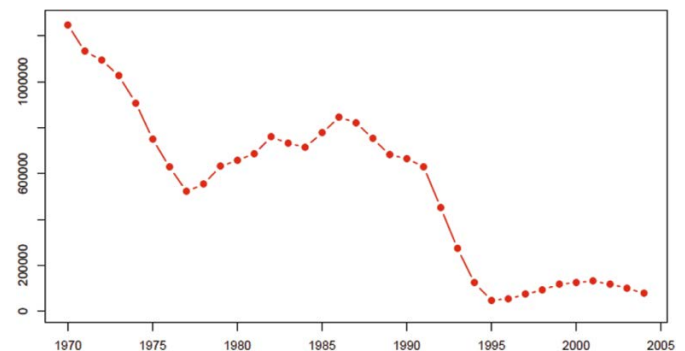
The predominant finfish species in aquaculture on both coasts is Atlantic salmon (Fig. 68). On the Atlantic, there is also a relatively small production of nonnative steelhead trout (Fig. 68A). While the nonnative Atlantic salmon currently dominates production on the Pacific coast, this was not always the case. Initially, the Pacific finfish aquaculture focused solely on native coho, Chinook, and steelhead trout (Fig. 68B). Atlantic salmon, however, was soon introduced into the Pacific aquaculture industry and, in 1987, the first harvest of Atlantic salmon occurred. Since then, the production of Atlantic salmon has expanded rapidly; today it constitutes about

Fig. 66. Landed value in thousands of 2011 dollars from various fisheries in the Atlantic (●) and Pacific (○) regions. Data source: DFO (2012).
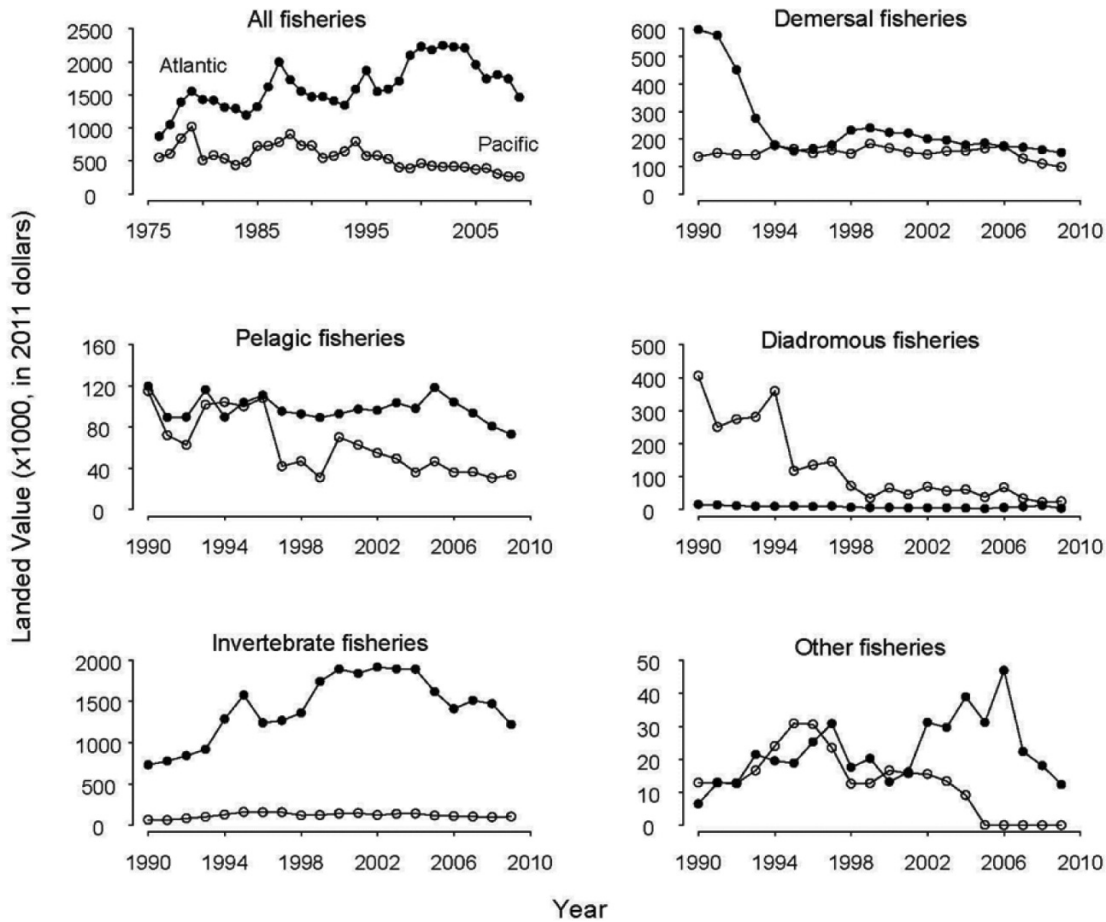
95% of BC's salmon aquaculture. BC is the 4th largest producer of farmed salmon in the world after Norway, Chile, and the UK. Moreover, its aquaculture production of salmon well exceeds that of its capture fisheries for salmon (78 700 and 23 100 tonnes, respectively, in 2010 (Government of British Columbia 2012). There are also two farms and one hatchery in BC that are rearing sablefish; the combined annual production of these facilities currently ranges between 2000 and 3000 tonnes.

Canadian aquaculture production of shellfish is about one-third, by weight, that of finfish (Fig. 64). In contrast to finfish aquaculture, the Atlantic region dominates production (~79% of Canada's production; Fig. 69). Both native and nonnative species are cultured in Canadian waters. The main producer in the Atlantic is Prince Edward Island (PEI), being responsible for about 62% of Canada's farmed shellfish. Shellfish aquaculture in the Atlantic involves primarily mussels (native species: *Mytilus edulis* and *M. trossulus*) and secondarily oysters (mainly native *Crassostrea virginica* with some nonnative *Ostrea edulis*) (Fig. 70). There are also small harvests of clams (native species: *Mya arenaria*, *Mercenaria mercenaria*), scallops (native *Placopecten magellanicus* and nonnative *Argopecten irradians* and *Chlamys islandica*), and abalone (nonnative *Haliotis rufescens*). By contrast, BC is Canada's major producer of oysters (nonnative species, primarily *Crassostrea gigas*, but also C. *virginica* and *O. edulis*), clams (nonnative *Nuttallia obscurata*

Fig. 67. Canadian aquaculture production of marine finfish. Production is dominated by salmonid fishes, particularly Atlantic salmon. Production of other marine finfish constitutes <1% of the total marine finfish aquaculture production. Data source: DFO (2012).
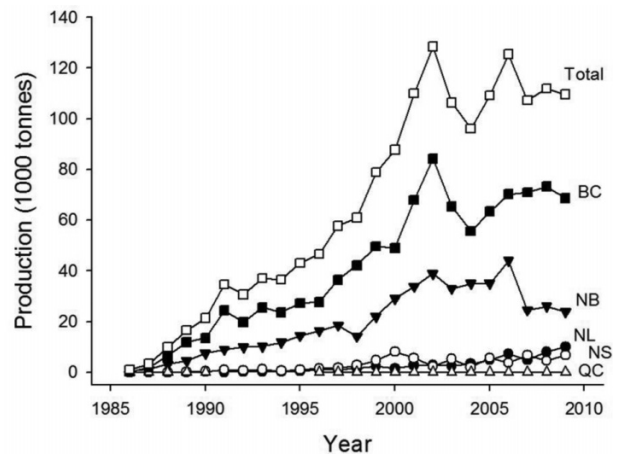
and *Tapes phillippinarum*, and to a lesser extent, the native *Protothaca staminea*), and scallops (nonnative hybrid *Patinopecten caurinus* X *P. yessoensis*). BC also has a small production of mussels (nonnative *Mytilus edulis* and *M.*

Fig. 68. Marine aquaculture production of salmonid fishes in (A) the Atlantic and (B) the Pacific regions of Canada. In 2003, species-specific production statistics for the Atlantic region were discontinued. In the Pacific region, production statistics could be separated into Pacific salmon and Atlantic salmon during 1986–1990 and 1998–2009, but not during 1991–1997. Data sources: DFO (2012) and Government of British Columbia (2012).

Fig. 69. Canadian aquaculture production of shellfish, by region. Source: DFO (2012).

6.3.2. Value of aquaculture production in Canada

In 2009, marine aquaculture production in Canada was valued at $736 million (Fig. 71). Of this, 69% was derived from exports, 97% of which was destined to the US. The value of marine finfish aquaculture was 10 times greater than that of shellfish; the value of aquaculture production in BC was ~26% greater than that in the Atlantic. In 2007, the total economic activity generated by aquacu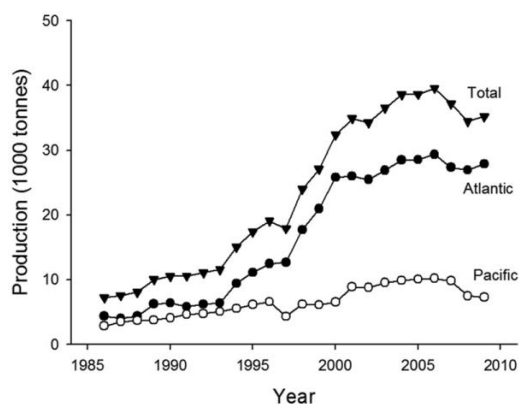lture in Canada was estimated at $2.1 billion (DFO 2010*e*); aquaculture was also estimated to account for 4895 direct jobs and 6400 indirect jobs (DFO 2010*e*).

6.4. Fisheries enhancement activities

Hatchery fish — artificially reared offspring of captively bred adults — are used for fish enhancement activities in the marine environment and are directed almost solely towards salmon. In BC, these activities are aimed towards supporting targeted fishing opportunities on enhanced stocks and, more recently, rebuilding severely depressed stocks. There has been a decline in the scale of these activities from a peak in the early 1990s, yet there remain roughly 110 facilities associated with DFO's Salmonid Enhancement Programme in BC. As of 2010, these facilities produced ~124 million Pacific salmon juveniles for release (Fig. 72A).

Canadian releases of hatchery salmon in the Atlantic are about two orders of magnitude less numerous than those in the Pacific (Fig. 72). There was a major shift in emphasis in Atlantic Canada in the mid-1990s, from one of production of surplus fish for catch, to that of conservation of vulnerable stocks. As a consequence, there was a decrease in enhancement releases (Fig. 72B). In 1997, DFO divested itself of eight of the nine hatcheries it had been operating in the Maritimes. The remaining hatchery subsequently became part of a "Live Gene Bank" (LGB) programme, mandated to conserve endangered fish populations. Later, two of the other hatcheries were eventually returned to DFO control under the auspices of the LGB programme. There are also a number of small-scale hatcheries run by provincial governments or nongovernmental organizations. Accompanying the change in focus to conservation, hatchery releases in the Maritimes since 2000 have become dominated by fry (the earliest feeding stage of salmon development) rather than older juveniles (i.e., parr and smolts). While the data for Québec are less complete, the pattern is similar to what has been documented in the

galloprovincialis). In 2009, shellfish culture involved 460 site tenures and all but two tenures were in southern BC; production totalled 7300 tonnes from a total area of 2114 ha (reported value of $16.3 million). Shellfish aquaculture on the west coast is expected to grow substantially; but at this time, it is much smaller than that on the east coast. There are also some additional invertebrate species under aquaculture development, including sea urchins (native *Strongylocentrotus droebachiensis* and *S. franciscanus*), geoducks (native *Panope abrupta*), abalone (native *Hailiotis kamtschatkana*), sea cucumber (native *Parastichopus californianus*), and cockle (native *Clinodardium nuttalli*).

Canada also has a small seaweed aquaculture industry, focused on native species, including Irish moss (*Chondrus crispus*), kelps (*Macrocystis integrifolia* and *Nereocystis luetkeanna*), brown algae (*Laminaria saccharina* and *L. groenlandica*), and knotted wrack (*Ascophyllum nodosom*). The principal provinces involved are BC, Nova Scotia, and PEI.

Fig. 70. Aquaculture production of (A) clams, (B) mussels, (C) oysters, and (D) scallops in Canada, by region. Data source: DFO (2012).
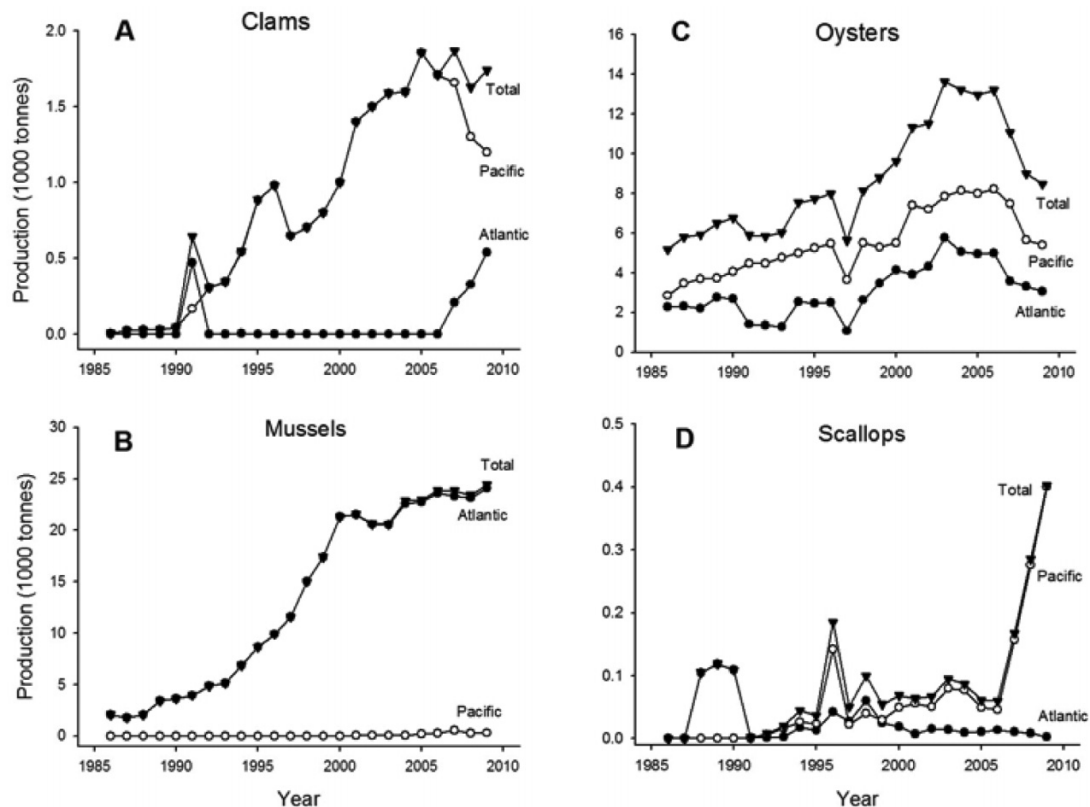
Maritimes (Fig. 72B). It is noteworthy that there have never been large releases of hatchery salmon in Newfoundland and Labrador.

7. Climate change: observed and projected consequences for Canadian marine biodiversity

7.1. Introduction

Climate change directly impacts key physical attributes of the oceans that, in turn, regulate biodiversity and ecosystem functioning. The immediate consequences of an altered climate can include changes in temperature, precipitation and hydrography, acidification, and sea level rise (section 4). Climate change will affect marine biodiversity. Direct effects are caused by the influence of physical and chemical factors, such as temperature, winds, vertical mixing, salinity, oxygen, and acidity that affect the physiology, development, reproduction, behaviour, and survival of organisms (Brander 2010). Indirect effects encompass those changes in communities that are mediated by food web interactions. Given the possibility that many such impacts may be more or less irreversible over long time scales (centuries), their effects will play a potentially critical role in managing marine biodiversity, especially with respect to ecosystem services (section 1), including Canada's fisheries.

These potential effects can be summarized in five principal categories: (*i*) patterns of net primary production and carbon export; (*ii*) biogeographical shifts that involve large-scale changes in the spatial distribution of organisms; (*iii*) phe-

nology and environmental mismatch, whereby changes in the oceans cause a mismatch between habitat requirements and resource availability; (*iv*) regime shifts, involving relatively rapid reorganisations of ecosystem functioning; and (*v*) biological responses to ocean acidification. These impacts, some of which are not mutually exclusive, are occurring in virtually all marine ecosystems, including the epipelagic, intertidal, benthic, and estuarine ecosystems (section 3), and are most evident at high latitudes. The section concludes with a consideration of how the effects of climate change might exacerbate the effects of fishing.

7.2. Net primary production and carbon export

Over the past 40 years, approximately 84% of the increase in Earth's heat budget has been absorbed by the ocean's surface waters. The concomitant sea-surface warming will likely affect the pelagic ecosystem in several ways: directly through its effects on the rates of biological processes and light supply responses to changes in cloudiness and mixed-layer thickness, and indirectly through decreased surface layer mixing and, hence, reduced nutrient supply (Sarmiento et al. 2004). Climate model simulations project an overall reduction of the density of surface waters due to warmer SSTs and fresher surface waters at high latitudes. This phenomenon is expected to increase both vertical stratification (and thus reduce nutrient input; section 4) and the length of growing seasons at high latitudes. However, a complex combination of factors leads to considerable geographical variation and considerable uncertainty in the predicted response of ocean pro-

Fig. 71. Value of (A) marine finfish aquaculture and (B) shellfish aquaculture in Canada separated by region. Data Source: DFO (2012).

Fig. 72. Releases of juvenile anadromous salmonids from enchancement facilities in (A) the Pacific region of Canada and (B) the Maritimes and Québec. Pacific salmon releases are divided into fry (recently emerged juveniles that are ready to feed) and smolt (age 1 or older juveniles ready to migrate to sea) releases. Data for the Pacific region were provided by DFO's Salmon Enhancement Programme, and for the Maritimes by DFO's Maritimes Region Science Branch. Data for Québec were obtained from annual *Bilan l'exploitation du saumon au Québec* reports of the Ministère des Ressources naturelles et de la Faune du Québec (RNQ 2012).

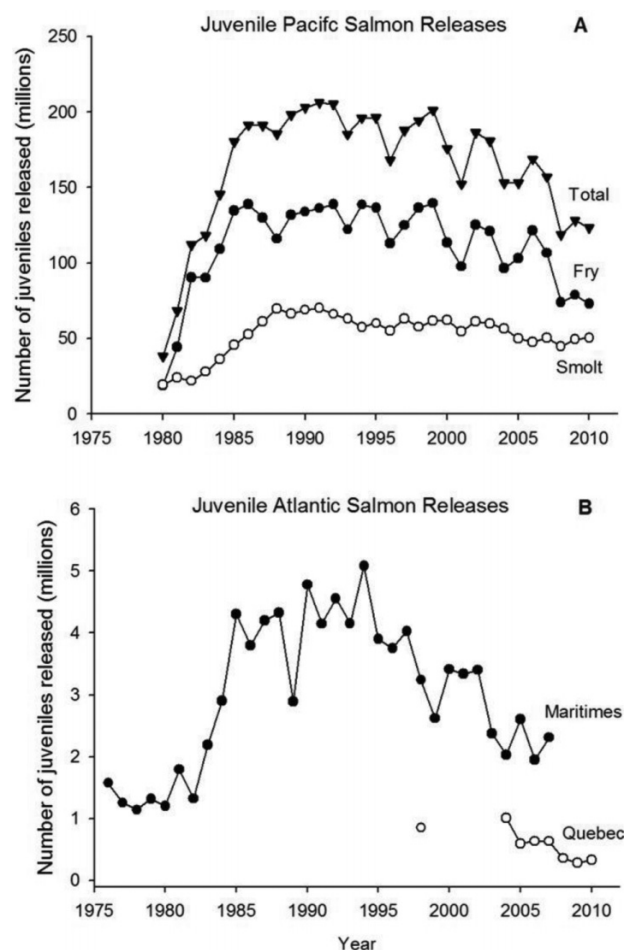ductivity to climate warming. According to some estimates (Sarmiento et al. 2004), primary production is projected to (*i*) increase globally between 0.7% and 8.1% by 2050; (*ii*) increase in the North Atlantic; and (*iii*) decrease in the North Pacific, due primarily to the retreat of the highly productive region of marginal sea ice. Other studies also predict a reduction in primary productivity. Steinacher et al. (2010), for example, projected a decrease in global mean primary productivity of between 2% and 20% by 2100 relative to preindustrial conditions, based on the outputs of four global-coupled carbon cycle – climate models.

The sensitivity of biological processes to temperature is described by the Q_{10} coefficient, namely, a measure of the rate of change of a biological process as a consequence of increasing the temperature by 10 °C. While phytoplankton photosynthesis and growth show a moderate response to increasing temperature (a Q_{10} of between 1 and 2), bacterial heterotrophic activities show a stronger response (a Q_{10} of between 2 and 3). In addition, bacterial growth efficiency declines with an increase in temperature, such that an increasing fraction of assimilated carbon is respired at higher temperatures. Based on such considerations, and taking into account experimental evidence, ocean warming may shift the balance between au-

totrophic production and heterotrophic consumption of organic matter toward enhanced recycling and respiration. Consequentially, the loss of carbon through sinking may be significantly reduced at higher temperatures. These changes have the potential to reduce the transfer of primary produced organic matter to higher trophic levels and weaken the biological carbon pump, thus providing a positive feedback to rising atmospheric CO_2 (Wohlers et al. 2009).

Consistent with these observations, pronounced latitudinal increases in phytoplankton biodiversity in parts of the North Atlantic Ocean are paralleled by a decrease in the mean size of phytoplankton (Beaugrand et al. 2010). In the Arctic Ocean, the smallest phytoplankton cells thrive as surface waters warm and freshen because of increased sea-ice meltwater and episodic input of fresh water from large-river runoff. Predictably,

Fig. 73. Projected zonal (latitudinal) changes (globally and by region) in 10-year average maximum fishery catch potential from 2005 to 2055 under high-range (black line) and low-range (grey line) greenhouse-gas emission scenarios. The dotted line indicates no change in catch potential. Source: Cheung et al. (2010).
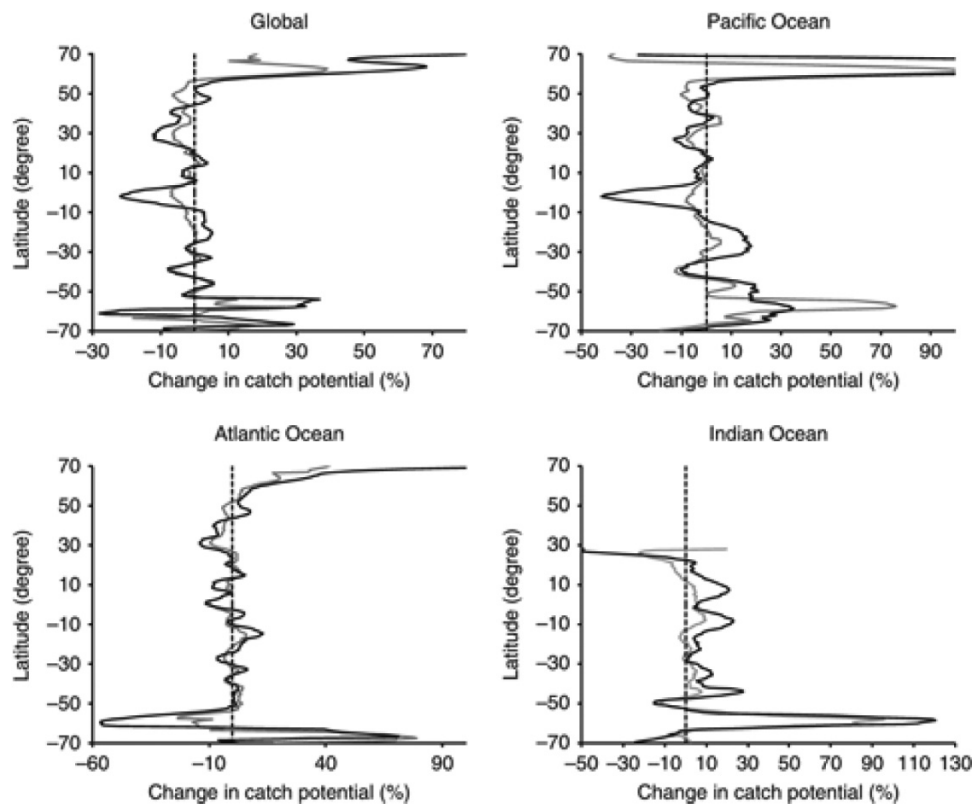
small picophytoplankton cells proliferate in a regime of lower nitrate supply and greater hydrodynamic stability (Li et al. 2009). A reduction in community average body size, caused by an increase in the abundance of individuals belonging to small-sized species, might represent a common response to global warming (Daufresne et al. 2009).

7.3. Biogeographical shifts

Environmental temperature plays a pivotal role in determining the spatial distribution of virtually all ocean plants and animals. By altering environmental temperature patterns, climate change affects marine biodiversity, and potential yields from fisheries, through changes in species distributions. One such example is the shift towards smaller organisms in North Atlantic plankton that has been thought to result in a reduced fisheries yield (Beaugrand et al. 2010).

Empirical and theoretical studies suggest that marine fishes and invertebrates respond to ocean warming by shifting their latitudinal and depth ranges, with observed and projected rates of range shift of between 30–130 km decade^{-1} pole-wards and 3.5 m decade^{-1} to deeper waters (Mueter and Litzow 2008; Cheung et al. 2009, 2010). Such changes may result in the local extinction of some species, the invasion of others, and increased rates of species turnover resulting in ecological modifications.

A globally based projection of the distributional ranges of a sample of 1066 exploited marine fish and invertebrates for the year 2050 (Cheung et al. 2009) indicates that climate change

may lead to an elevated level of local extinction in subpolar regions, the tropics, and in semienclosed seas, resulting in species turnovers of more than 60% of the present biodiversity. In combination with projected patterns of primary production, the catch potential in global fisheries will in all likelihood undergo significant changes (Cheung et al. 2010; Fig. 73). Globally, in the northern hemisphere, 10-year average changes in maximum catch potential (projected from 2005 to 2055) indicate a moderate decline in temperate regions (25°N–50°N), but increases at higher latitudes, particularly in the Subarctic (Fig. 73, upper left panel). In the Pacific Ocean, the pattern of change in catch potential parallels the global trend, but with a much higher magnitude of change (Fig. 73, upper right panel). In the Atlantic, the projected magnitudes of change in temperate regions are smaller than those in the Pacific (Fig. 73, lower left panel). It is important to note, however, that these projections do not account for potentially strong ecological interactions among species (see below), nor do they account for physiological impacts on fish metabolism, such as reduced growth performance resulting from increased acidification and lower oxygen content (Cheung et al. 2011b).

In Canadian waters, warming ocean temperatures are predicted to result in the loss of some fish and invertebrate species but a gain in others (Fig. 74; Cheung et al. 2011a). The greatest species losses (6–10 species) are predicted to occur at lower latitudes, including the Scotian Shelf, the Newfoundland-Labrador Shelf, and the marine ecoregions of the Pacific. Overall, how-

Fig. 74. One forecast of how changes in water temperature between 2005 and 2050 might affect the distribution of fish and invertebrate species in Canadian waters. (A) Predicted changes in SST. (B) Predicted turnover in number of species (species gains minus species losses). (C) Predicted proportional change in numbers of species. Source: Cheung et al. (2011*b*).
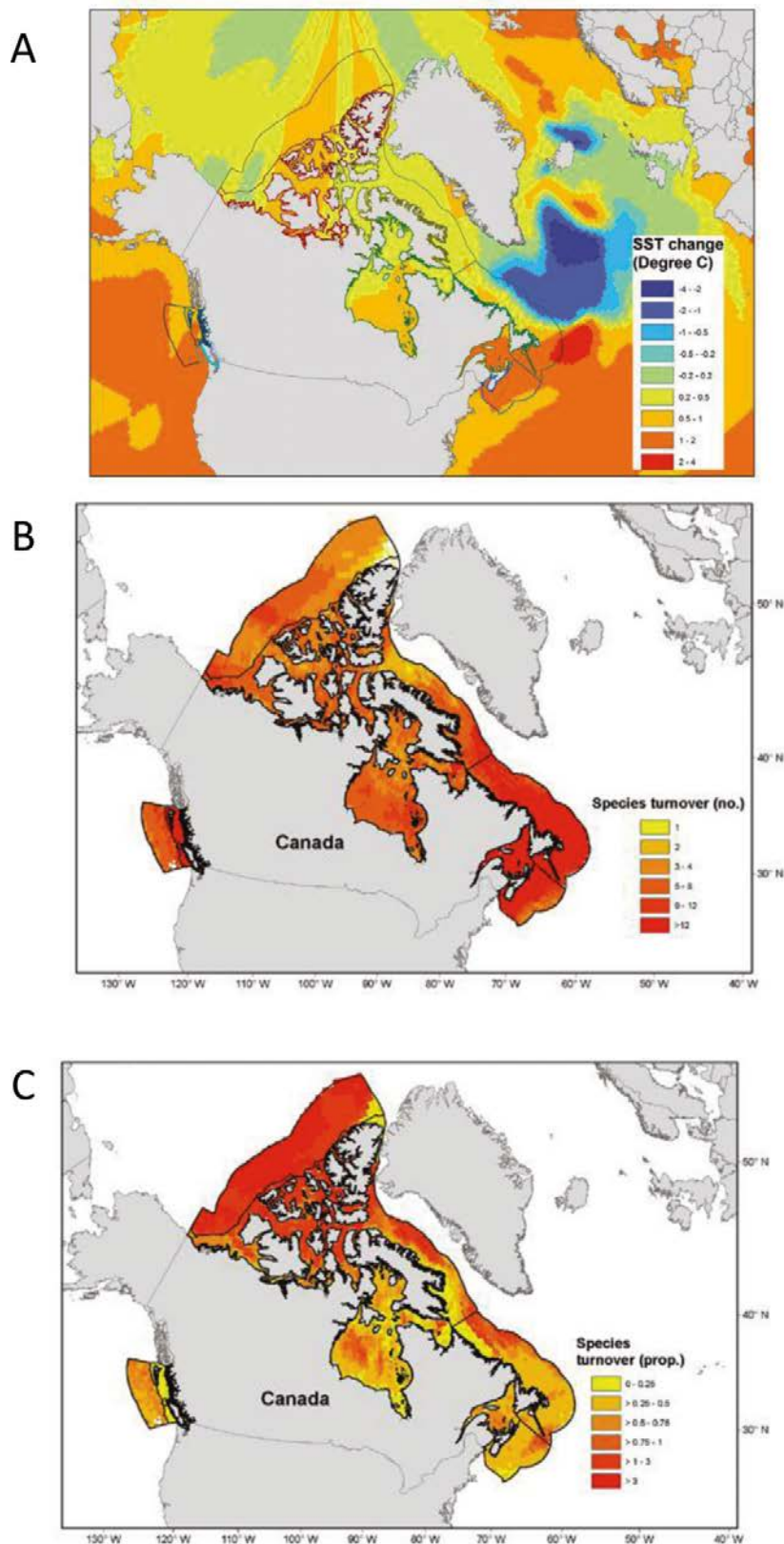
ever, the predicted pattern is one of species gain. Species turnover (species gains minus species losses) is predicted to be greatest at lower latitudes and throughout the Arctic; as a proportion of the current numbers of species, species turnover might be highest in the western Arctic (Fig. 74).

A striking example of relatively rapid changes in the biogeography of the epipelagic ecosystem in response to climate warming is provided by the copepod communities of the Northeast Atlantic and European shelf seas (Beaugrand et al. 2002). Major biogeographical shifts for all species assemblages have occurred since the early 1980s, with a northward extension of more than 10° of latitude for warm-water species. In contrast, the diversity of colder-temperate, Subarctic, and Arctic species has declined. All the biological associations show consistent long-term changes, reflecting a shift in marine ecosystems towards a warmer state, with potential modifications in the abundance of exploited boreal species, such as Atlantic cod.

Biogeographical shifts in species distribution have already affected intertidal biodiversity and community structure. The locations of range edges of many rocky intertidal species found in the North Atlantic, for example, have shifted by as much as 50 km decade^{-1}, much faster than most recorded shifts of terrestrial species. The rate and extent of contractions in the equatorial range limits are less than the changes observed at the pole-ward limits of distribution. These biogeographic shifts have been accompanied by increases in the abundance of many species close to their pole-ward range limits and by changes in the relative abundance of warm- and cold-water species (Helmuth et al. 2006).

One means by which a warmer ocean can produce indirect effects on biodiversity is through temperature-mediated ecological interactions. For example, the upper limits of zonation of many species of intertidal invertebrates and algae are correlated with maximum air temperatures. When these upper limits of an organism are compressed down to the upper limit of a predator or dominant competitor, the prey or subordinate species is eliminated from the intertidal zone (Helmuth et al. 2006). Such species interactions may have a greater impact on community structure than simply the replacement of cold-water species by warm-water species. A 3.5 °C rise in seawater temperature, induced by the thermal outfall of a power-generating station, over 10 years along 2 km of rocky coastline in California resulted in significant, but largely unpredictable, community-wide changes in 150 species of algae and invertebrates (Schiel et al. 2004). Temperature-sensitive algae (particularly subtidal kelps and intertidal red algae) decreased in abundance while many invertebrate grazers increased in number. Community response was, thus, coupled to the direct effects of temperature on key species and the indirect effects operating through ecological interactions (Schiel et al. 2004).

The ecological consequences of species' range expansions, particularly those of exotic species invading new habitats, are of concern in the context of a warmer ocean. Although only a small fraction of the many marine species introduced outside of their native range are able to proliferate and invade new habitats, their effects can be profound.

An increase in the number of marine invasive species fostered by a warmer ocean may have a negative impact on marine biodiversity. One example from the Mediterranean Sea serves to illustrate this phenomenon. A large part of the eastern

Fig. 75. Proportional representation of Arctic cod, *Boreogadus saida,* and capelin (% loads delivered) fed to nestling thick-billed murres at Coats Island (northern Hudson Bay) between 1981 and 2007 (no data for 1982 and 1983). Source: Gaston et al. (2009).
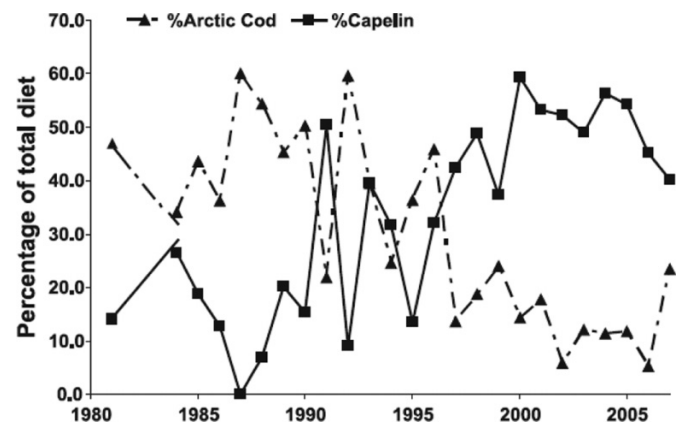
Mediterranean Sea is undergoing substantial warming which has modified the potential thermal habitat available for warm-water species, facilitating their establishment at an unexpectedly rapid rate (Raitsos et al. 2010). There has been a 150% increase in the annual mean rate of species entry since 1998. The speed of alien species spreading, a response to global warming, appears much faster than temperature increase itself, which could lead to a restructuring of the pelagic ecosystem (Raitsos et al. 2010). Whether such changes are judged in the future as positive, negative, or neutral, remains to be seen.

Closer to home, and with unpredictable consequences, the warming of the Arctic Ocean is expected to facilitate the expansion of Pacific species into a warmer Arctic Ocean, and eventually into the North Atlantic Ocean. Trans-Arctic invasions began about 3.5 million years ago, having been periodically halted by sea-ice expansion in the coastal Arctic Ocean. Conditions are ripe, however, for invasions to resume, with at least 77 molluscan lineages having the potential to extend to the North Atlantic via the warmer Arctic Ocean without direct human intervention (Vermeij and Roopnarine 2008).

Recent evidence suggests that new invasions have begun, probably facilitated by modified current regimes (section 4). In 1999, a long-term monitoring programme documented the presence of a Pacific diatom between Canada and Greenland. The species, known previously only from the North Pacific and Bering Sea, has subsequently spread south to Georges Bank and east to Iceland, providing one indication of the speed and scale of change that can take place in response to climate warming (Reid et al. 2008). Capelin (*Mallotus villosus*), an important forage species in northern waters, has recently spread from the western Arctic eastwards to Cumberland Sound (Dodson et al. 2007), a distributional shift that might explain the proportional shift in diet in at least one species of seabird (thick-billed murre, *Uria lomvia*) in the eastern Arctic (Fig. 75; Gaston et al. 2009).

7.4. Phenology and environmental mismatch

Changes in temperature directly impact the timing of life-history events (phenology) that comprise the life-cycle of marine organisms. These events include, among others, reproduction,

Fig. 76. Trend in the timing of peak biomass of the zooplankton, *Pseudocalanus neoplumchrus*, in the Strait of Georgia (Julian date = day of year). Source: DFO (2010*c*).

Fig. 77. Mean Julian dates (day of year) of hatching for four species of seabirds at Triangle Island, BC. Source: Gaston et al. (2009).
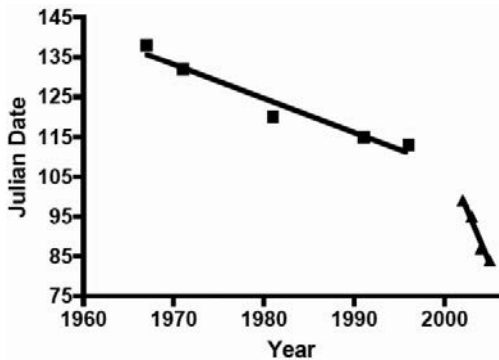
hatching, and metamorphosis. Climate plays a critical role in controlling the match between predator requirements and resource availability (Cushing 1969; Visser and Both 2005; Durant et al. 2007). Climate change affects the relative timing of food requirement and food availability for various organisms and, by doing so, influences their reproduction and survival. Differences in the temporal and spatial match between predator and prey thus generate variability in predator survival rates.

One striking example of a change in the timing of life-history events concomitant with temperature change is the pattern of increasingly earlier seasonal timing of the peak abundance ("bloom") of the dominant species of zooplankton in the Strait of Georgia, *Pseudocalanus plumchrus* (DFO 2010*c*). In the late 1960s, it has been estimated that the spring bloom occurred in mid-May (Fig. 76). By 2004, the bloom was occurring in mid-March, a trend associated with warming SSTs in this area (section 3). The progressively earlier timing of peak abundance will have some effect on the multitudes of species that utilize this zooplankton for food. It may, for example, partially explain the earlier hatch dates observed in some Pacific seabirds (Fig. 77), for which the consequences to the persistence of these species are not known.

In a survey of 25 532 rates of phenological change for 726 UK terrestrial, freshwater, and marine taxa, the majority of spring and summer events were found to be occurring earlier, and more rapidly, than previously documented (Thackeray et al. 2010). Phenological events associated with predators advanced less rapidly than those for prey species. Such consistency is indicative of shared large-scale drivers across terrestrial, freshwater, and marine environments. There is, thus, compelling evidence that climate-change-driven phenological asynchronies can disrupt the stability and functioning of ecosystems and, as a consequence, the delivery of key ecosystem services.

Temperate and high-latitude epipelagic ecosystems might be particularly vulnerable to phenological changes caused by climatic warming (Rubao et al. 2010). Recruitment success at higher trophic levels is dependent on synchronization with seasonally pulsed primary production; the response to regional warming varies among functional groups. For example, changes in the North Sea planktonic assemblage and copepod phenology were correlated with warming over the last few
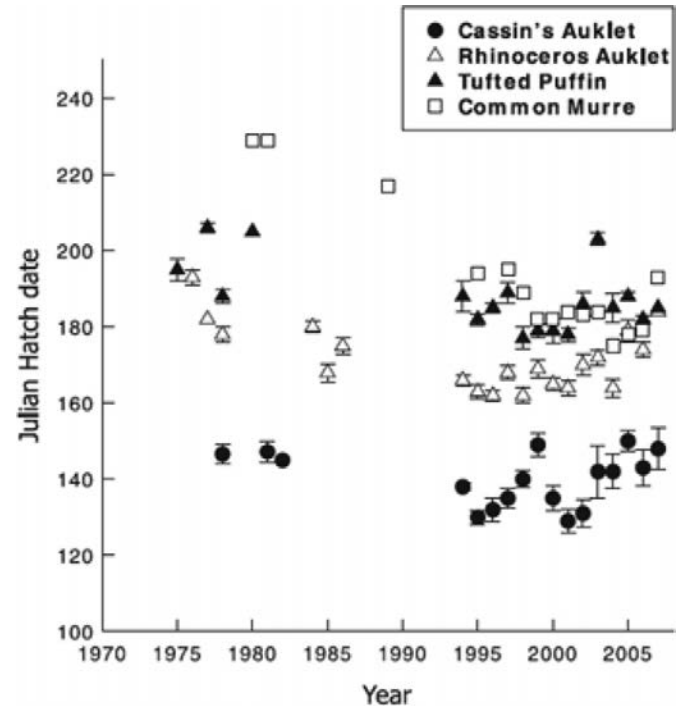
decades. This has resulted in a poor food environment for cod larvae and a decline in overall recruitment success (Beaugrand et al. 2002). Similar evidence linking plankton phenology and higher trophic levels has been found elsewhere in the North Atlantic (phenology of shrimps and phytoplankton; Koeller et al. 2009) and the North Pacific (seasonality of zooplankton abundance and energy propagation to fish and seabird predators (Mackas et al. 2007; Bertram et al. 2009). A potential mismatch between the primary production peaks of high-quality food and the reproductive cycle of key Arctic grazers may have negative consequences for the Arctic epipelagic marine ecosystem (Soreide et al. 2010).

Environmental mismatch can also occur between distinct life-history stages and the environmental conditions encountered during ontogeny. This ontogeny–climate interaction occurs when the physical tolerances and habitat requirements of individuals change through development and because individuals may migrate among habitat types. Species that undergo long-distance migrations are especially prone to this effect because their migrations take them to geographic regions with distinct climates. For example, the timing of the downstream migration of Atlantic salmon smolts in Gulf of St. Lawrence populations appears to be out of synchronization with ocean conditions in the postsmolt nursery areas; decreased smolt survival is associated with abnormally warmer SSTs (Friedland et al. 2003). Another example is provided by Chinook salmon. The stream-type Chinook salmon life history is, like many Pacific salmonids, characteristic of cooler water. The entire migration and spawning strategy is adapted to a snowfall-dominated hydrological regime typical of Pacific Northwest rivers. However, global warming is predicted to increase summer stream

temperatures and reduce summer/fall flow levels. Temperatures that are above optimal levels impede developmental processes (e.g., smoltification), predator-avoidance behaviour, and growth (Crozier et al. 2008). Extremely low flows may decrease survival in small streams by reducing potential habitat availability, thereby increasing both competition for food and predator mortality. The mismatch created by climate impacts on juvenile habitat requirements is projected to reduce population abundance and significantly increase the probability of extinction for affected populations (Crozier et al. 2008).

Nowhere is the mismatch between the requirements of developmental stages and the availability of adequate habitat more striking than in the Arctic and subpolar regions of both the Pacific and Atlantic oceans (e.g., Gulf of St. Lawrence region, Labrador Sea) (section 4). Reductions in the extent of sea ice will be most harmful to species that rely on ice as a platform for breeding or foraging (Friedlaender et al. 2010). Ice-related declines in abundance have been reported in hooded seals (*Cystophora cristata*), harp seals, and ringed seals (*Pusa hispida*), and there is also evidence of impacts on the distribution and health status of Pacific walrus (*Odobenus rosmarus*) (Vincent et al. 2011). Early ice breakup may result in premature separation of mothers and pups, leading to higher mortality rates among pups. In the southern Baltic Sea, from 1989 to 1995, a series of nearly ice-free winters led to very high pup mortality rates (Härkönen et al. 1998). Similarly, several nearly ice-free winters in the Gulf of St. Lawrence have led to high mortality rates among harp seal pups. Polar bears (*Ursus maritimus*) are especially prone to the negative consequences of changing sea-ice conditions and severe population declines have been projected for some areas of the Arctic (Regehr et al. 2010).

Although its effect may not strictly be a case of mismatching of ecological processes, changes in SST are known to be associated with changes in productivity (i.e., adults produced per spawner) of Pacific salmon populations. The relationship is either positive or negative, depending on the location along North America's west coast. Specifically, increased SST in the location of early ocean residence of a given population's juveniles is associated with increased productivity in Alaska, but *decreased* productivity for Washington State and most BC pink and sockeye salmon (Mueter et al. 2002). The highest SST values are well below the lethal temperature limit for salmon, so it is thought from various sources of evidence that SST is simply an indirect indicator of the food supply for juvenile salmon and the abundance of their predators.

7.5. System regime shifts

Perturbations in the marine realm can cause ecosystems to shift between contrasting, sometimes persistent states (deYoung et al. 2008), with major alterations to community composition and ecosystem services (e.g., Grebmeier et al. 2006). Such shifts can occur quite suddenly because ecological relationships such as predation and competition between key species are affected. The major drivers of ecosystem regime shifts include abiotic factors, such as changes in ocean stratification, storm events, and temperature, and biotic factors such as community changes resulting from overfishing or species invasions. Four examples below serve to illustrate how marine ecosystems can shift to drastically different forms and functions in response to changing climate.

In the North Pacific, there have been large and rapid changes in ocean productivity, with the greatest change occurring as a result of intensification of the Aleutian low-pressure atmospheric system over the winter of 1976–1977 (Mantua et al. 1997). That atmospheric forcing changed wind and current patterns in such a way that the eastern North Pacific Ocean became more productive than in the past. Hare and Mantua (2000) found clear evidence of biological responses to these climatic changes in their analysis of 31 climatic and 69 biological time series, the latter of which included data series as diverse as zooplankton, shrimp, demersal fish, pelagic schooling fishes, and salmon. For instance, catches of sockeye salmon increased after 1977, especially in Alaska, a result that was consistent with data on a biological measure of productivity (adult returns per spawner) (Peterman et al. 1998). It is thought that increased spin-up of the Gulf of Alaska gyre, starting in 1976–1977, brought more nutrients to the surface, leading to increased primary and secondary productivity. Hare and Mantua (2000) also found another regime shift in 1989, but that shift was less extensive and did not return to pre-1977 conditions. Mueter et al. (2007) confirmed that both pelagic productivity (mostly salmon) and demersal productivity increased in response to the 1976–1977 climatic regime shift, whereas the 1988–1989 regime shift produced inconsistent or short-lived responses.

On a smaller scale, in 1987, the benthic community on rocky reefs of southern California underwent an abrupt shift from a community in which sea cucumbers were rare to one in which they dominated (Rassweiler et al. 2010). The new community state persisted until 2002 when sea cucumber densities suddenly declined dramatically everywhere, remaining relatively low through 2008. It seems that a combination of an unusual period free of large storm events and low abundance of seastar predators allowed sea cucumber populations to increase dramatically. The shift back to few sea cucumbers coincided with an increase in the number of predatory seastars. The consequences for primary production of high sea cucumber numbers were profound. The abundance of macroalgae, which compete for space with sea cucumbers, plummeted as the sea cucumber population grew, with cascading effects reverberating through the entire food chain in the form of reductions in herbivorous microcrustaceans and their associated fish predators (Rassweiler et al. 2010).

Additional examples from coastal waters serve to illustrate how multiple factors can interact to generate synergistic responses, which occur when the impact of combined factors is disproportionately amplified compared with that of those same factors operating individually. Kelp forests occur along the majority of the world's temperate coastlines. On many coasts where humans have altered chemical and biological conditions, however, kelp forests have been replaced by mats of turf-forming algae. While kelp canopies inhibit turfs, shifts from kelp to turf domination are fostered by reduced water quality, enabling the cover of turf to expand and persist beyond its seasonal limits. Controlled experiments have shown that, at elevated levels of CO_2 and temperature, such as those expected by 2050, algal turfs grew very quickly — much more quickly than under each elevated condition separately or under present-day conditions (Connell and Russell 2010). Moreover, turfs inhibited kelp recruitment, potentially locking the ecosystem into a turf-dominated state in the long term

(Connell and Russell 2010). Johannessen and MacDonald (2009) provided an excellent case study of the effects of climate change on a small spatial scale (Strait of Georgia) in Canadian waters.

7.6. Hypoxia

In Canadian waters, both the Pacific and Atlantic coasts are sites of growing hypoxic bottom waters (see section 4). A water mass is hypoxic when the concentration of dissolved oxygen is so low as to cause stress for aquatic organisms. In general, this occurs when the oxygen concentration is <2 mg L^{-1} (although some fish larvae may suffer at <3 mg L^{-1} and other organisms such as euphausiids can survive to 0.1 mg L^{-1}; Ekau et al. 2010). At this level, many fish species cannot survive, and the benthic community structure undergoes significant modifications. The dominant natural processes involved in the formation of hypoxic waters are photosynthetic carbon production and microbial respiration. The resupply rate of oxygen is indirectly related to its isolation from the surface layer. Hypoxic water masses thus occur at depth and are more likely to occur in systems where the water residence time is extended, water exchange and ventilation are minimal, stratification occurs, and where carbon production and export to the bottom layer are relatively high (Rabalais et al. 2010).

The formation of hypoxic areas is exacerbated by any process that increases primary production and the accumulation of organic carbon, leading to increased respiratory demand for oxygen below a pycnocline (see section 4). Thus, nutrient loading and coastal eutrophication are particularly problematic. Climate change threatens to further complicate the situation. The likelihood of strengthened stratification, stemming from increased surface water temperature as the global climate warms, is sufficient to exacerbate hypoxia where it currently exists and to facilitate its formation elsewhere. Heavier precipitation that increases freshwater discharge and the flux of nutrients will contribute to increasing local primary production, thus promoting hypoxia (Rabalais et al. 2010). Along the Oregon coast, for example, low-oxygen events have caused fish and crab kills at the ocean bottom during the last several years, events that had not been observed in the previous century (DFO 2011b).

7.7. Ocean acidification

When the CO_2 concentration increases, more carbonic acid (H_2CO_3) is formed, which partially dissociates into bicarbonate (HCO_3) and hydrogen (H+) ions, resulting in increased acidity as reflected by lower water pH. The pH is a measure of H+ activity and is an important water quality indicator because fish and other organisms are highly sensitive to changes in pH. This ocean acidification decreases the availability of carbonate, making it more difficult for many marine organisms to construct their hard parts out of calcium carbonate minerals. The combination of increased acidity and decreased carbonate concentration also has implications for the physiological functions of numerous marine organisms. When seawater is under-saturated in calcium carbonate ($CaCO_3$), structures composed of carbonate tend to dissolve. In such cases, seawater is corrosive to $CaCO_3$ (Fabry et al. 2008, 2009). The pH of the oceans is declining rapidly. Models project that surface waters of the Arctic Ocean and parts of the Subarctic Pacific will become increasingly corrosive to $CaCO_3$ as early as the mid-

dle of the 21st century (Fabry et al. 2009). As noted previously in section 4, because of the projected release of fresh water to the North Atlantic over the next century, waters as far south as Newfoundland may become under-saturated with respect to aragonite by 2100 (Denman et al. 2011).

Many marine organisms that use calcium carbonate to construct their shells or skeletons — including corals, coccolithophores (calcareous phytoplankton), lobsters, mussels, snails, and sea urchins — are potentially the most vulnerable to acidification. As carbonate becomes scarce, organisms should find it increasingly difficult to form their skeletal material. For example, decreased calcification rates will slow the growth of coral reefs and make them more fragile and vulnerable to erosion. By the middle of the 21st century, reef erosion may surpass reef-building. Many laboratory studies on a variety of coral species, indeed almost every study published to date (reviewed by Doney et al. 2009), confirm that coral calcification rates decrease in response to decreasing aragonite saturation state. Analyses of cores from coral colonies of the Great Barrier Reef show that calcification rates declined 21% between 1988 and 2003 (Cooper et al. 2008). (Canada's cold-water corals may respond similarly to those in south-temperate and tropical waters; research in this area, however, is lacking.)

Responses of other groups of calcifying organisms to ocean acidification may be more variable (Ries et al. 2009). In the laboratory, blue mussels exhibit no response to variation in CO_2 levels, while the calcification rate of limpets, urchins, coralline red algae, and calcareous green algae increases at 2–3 times preindustrial levels, before declining at higher levels. In three species of crustaceans (crabs, lobsters, shrimps), net calcification rate was greatest under a CO_2 level equivalent to 10 times preindustrial levels (Ries et al. 2009).

The potential impacts of ocean acidification appear to be far reaching and complex, even for noncalcifying species such as fishes. Fish larvae exposed to high CO_2 concentrations display odd behaviours, such as an attraction to the smell of predatory fish and a dangerous willingness to leave shelters, resulting in dramatic reductions in survival and recruitment (Munday et al. 2010). At the larval stage of development, fish might not have developed appropriate mechanisms for acid–base regulation. Increased levels of CO_2 have been shown to be associated with reduced growth and increased mortality in inland silversides (*Menidia beryllina*; Baumann et al. 2012) and severe damage to internal organs in Atlantic cod (Frommel et al. 2012). On a somewhat positive note, ocean acidification can cause increases in carbon fixation rates in some photosynthetic organisms (both calcifying and noncalcifying), potentially contributing to the efficiency of the carbon pump. In general, however, the potential for marine organisms to adapt to increasing CO_2, and the broader implications of acidification for marine biodiversity and ecosystem functioning, are not well understood.

7.8. Interactions between climate change and fisheries

Climate is influencing populations and communities that are also affected by fishing. Both climate and exploitation contribute to observed trends and interact in their effects. Fishing, by altering the structure of populations and ecosystems, can modify their response to climate (Planque et al. 2010; Shackell et al. 2010). The demographic effects of fishing that lead to changes in age or size structure and distribution (section 8) can

modify the capacity of populations to buffer climate effects. Similarly, changes in the relative abundance or loss of population subunits may lead to a reduction in the capacity of the wider population to tolerate climate variability and change (Planque et al. 2010). Although detailed knowledge of the role of climate in influencing fisheries productivity, and our capacity to predict the effects of future climate change, is relatively limited, existing knowledge is sufficient to advise on many aspects of management in a changing climate. Broadly, the lower rates of fishing mortality associated with maintaining biomass at or above MSY will help provide populations with greater resilience to climate change, as will any management measure that preserves a broad age composition (e.g., Brander 2010). Given that climate change is occurring and will continue to occur, management systems will need to be appropriately responsive to this key stressor on marine biodiversity.

8. Fisheries: observed and projected consequences for Canadian marine biodiversity

8.1. Introduction

Fisheries can have multiple consequences for marine biodiversity. The most direct means is through reductions in the numbers of individuals directly targeted as catch by a fishery or caught incidentally, as bycatch. Such an effect need not, however, be problematic from a biodiversity perspective. It depends on the extent to which the population is reduced, relative to the levels at which it is predicted to be sustainable in the long term, both from a single-species and multispecies (or "ecosystem") perspective. However, populations that decline further might experience reduced probabilities of persistence and contribute differently to ecosystem functionality. The deployment of fishing gear can reduce marine biodiversity through physical impacts on nontargeted individuals and the modification or destruction of their habitat. Also, by affecting the abundance of some species, fisheries have the potential to significantly influence interactions among others. Reduced abundance of a predator or group of predators, for example, can result in substantial increases in the abundance of their prey. Conversely, reductions of a fished species, concomitant with unchanged or even increased abundance of one of its predators, can significantly retard, or even prevent, the recovery of the depleted species.

The recovery of populations and marine biodiversity can also be influenced by fisheries-induced changes to the life-history traits (e.g., age and size at sexual maturity, growth rate) of the depleted population, i.e., characteristics that directly influence individual reproductive success and, thus, population growth rate. Some life-history trait changes might represent genetic or evolutionary responses to exploitation, meaning that, in the absence of fishing, they will not revert very rapidly to their former states. There is also evidence that fishing-induced reductions in the breadth or range of body sizes and ages in a population can also affect recovery potential.

8.2. Reductions in abundance

Sustained exploitation generally reduces population abundance and always reduces population biomass (i.e., the total weight of all individuals). For many Canadian species, these reductions have been among the greatest recorded for fish

Table 4. Marine fish populations estimated by COSEWIC to have declined more than 80% since the 1960s–1970s and for which overfishing has been identified as a cause for the decline (data obtained from COSEWIC species status reports; COSEWIC 2012a).

Species	Population (estimated magnitude of decline)
Atlantic Ocean	
Atlantic cod	Newfoundland & Labrador, including Grand Bank (97%), Northern Gulf and Newfoundland South Coast (89%), Southern Gulf & Eastern Scotian Shelf (90%)
American plaice	Gulf of St. Lawrence (86%), Newfoundland & Labrador (96%)
Northern wolffish	>95%
Spotted wolffish	>90%
Winter skate	Southern Gulf (98%), Eastern Scotian Shelf >90%)
Roundnose grenadier	98%
Porbeagle	89%
Deepwater redfish	Gulf of St. Lawrence (98%), Newfoundland & Labrador (98%)
Acadian redfish	99%
White shark	80%
Pacific Ocean	
Basking shark	>90%
Canary rockfish	78%–96%
Bocaccio	85%–90%

worldwide, particularly in the Atlantic, where total biomass of species such as Atlantic cod, American plaice (*Hippoglossoides platessoides*), Acadian redfish (*Sebastes fasciatus*), roundnose grenadier (*Coryphaenoides rupestrus*), and winter skate (*Leucoraja ocellata*) have declined by more than 90% since the 1960s (www.cosewic.gc.ca; accessed 18–11–2011). Most declines were experienced by species targeted by fishing, although some, such as winter skate, declined primarily because they were frequently caught as bycatch. Among marine and diadromous marine fishes assessed by COSEWIC as of January 2012, and for which fishing was the primary cause of decline, 18 are estimated to have declined more than 80% since the 1960s and exhibited little recovery (Table 4). The biodiversity of Canadian sharks has also been reduced by fishing. Notable examples, all of which have been assessed by COSEWIC as species at risk, include basking shark (*Cetorhinus maximus*), porbeagle (*Lamna nasus*), white shark (*Carcharodon carcharias*), shortfin mako (*Isurus oxyrhinchus*), bluntnose sixgill shark (*Hexanchus griseus*), blue shark (*Prionace glauca*), and spiny dogfish (*Squalus acanthias*).

Past overexploitation reduced the abundance of many marine mammals and led to the extirpation of at least one species from Canadian waters (grey whale, *Eschrichtius robustus*). However, populations of many of these species have since grown following a cessation, or considerable reduction, in exploitation. For example, after being subjected to commercial whaling from 1500 until 1910, the Arctic's bowhead whale (*Balaena mysticetus*) has increased in abundance, having been subjected to only sporadic hunting by Inuit during the past

Fig. 78. Trawl contents during the northern shrimp survey in Northwest Atlantic Fisheries Organization Division 0B (2006, 2008), showing catches of corals and sponges. Photo credit: DFO.

century. Both grey and harp seals in Atlantic Canada have shown extraordinary population growth rates following reductions in hunting pressure in the 1960s and 1970s (DFO 2010c). By the 1900s, the sea otter (*Enhydra lutris*) had been extirpated from BC by the fur trade. The species was reintroduced from 1969 to 1972 and has since repopulated 25%–33% of its historic range in Canadian Pacific waters.

Although quantitative historical data on marine bivalves are rare, it is clear that exploitation can, and has, resulted in significant depletions of some species in some areas. One example on Canada's east coast is the depletion of natural oyster beds in the southern Gulf of St. Lawrence (MacKenzie 1996; Milewski and Chapman 2002). On the west coast, harvesting has reduced northern abalone (*Haliotus kamtschatkana*) to such an extent that it has been assessed as endangered by COSEWIC.

8.3. Effects on benthic communities and habitats

Fishing activity is patchily distributed. This reflects decisions by fishers, who are influenced by past catch rates, potential catching opportunities, the cost of fishing, interactions with other fishers, regulations or incentives in the management system, and competition for space with other users of the sea. Location and gear choice influences the frequency and intensity of disturbances that affect different habitat types. Almost any fishing gear will disturb marine habitat to some degree. The response of habitats to fishing depends on their sensitivity and the type and intensity of fishing effects. In general, towed fishing gears that directly impact the seabed, such as trawls and dredges, are responsible for most fishing-related alteration or destruction of habitat (Fig. 78).

Meta-analyses of the recovery times of different habitats following fishing disturbance show that the impacts of scallop-dredging and other towed bottom-fishing gears on biogenic habitats (i.e., habitats produced or brought about by living organisms) are the most significant (Collie et al. 2000; Kaiser et al. 2006), while the biota of soft-sediment habitats, in particular muddy sands, can recover in a matter of years. Smaller free-living species with short lifespans tend to recover more quickly, especially in areas where they are already adapted to relatively high levels of natural disturbance.

Glass sponge (family: Hexactinellidae) communities or reefs provide a noteworthy example of sensitive habitat. These are typically found in the deep sea (at more than 650 m depth) but are also found in shallow waters (in as little as 16 m) in the Queen Charlotte Basin, Howe Sound, Strait of Georgia, and fjords of BC (e.g., Leys et al. 2004; Conway et al. 2005; Marliave et al. 2009; Chu and Leys 2010). Following early reports of the presence of sponge colonies and reefs, high-resolution habitat mapping has better described their distribution on the western Canadian continental shelf (Conway et al. 2001, 2005). Glass sponge reefs provide habitat for species such as rockfishes, *Sebastes* spp. (Marliave et al. 2009; Chu and Leys 2010); they "process" significant quantities of water (a recent estimate suggested they could remove 0.96 g C m^{-2} day^{-1}; Chu and Leys 2010); and they are sensitive to any impact from fishing gear (Conway et al. 2001; Cook et al. 2008). Other sensitive habitats in BC waters include "gardens" of sea whips (order: Gorgonacea) that have been impacted by shrimp beam trawls (Troffe et al. 2005) and cold-water marine ecosystems dominated by sponges in the Northwest Atlantic Fisheries Organization (NAFO) convention area off the Atlantic coast.

Trawling and dredging for fish and shellfish are widespread and locally intense. Their impacts on a range of habitats have been described. In one of the first studies, Messieh et al. (1991) estimated that the length of track swept by trawlers and dredgers fishing off Canada's east coast was 4.3 million km^2 in 1985. Experimental studies of the effects of otter trawling on the hard-bottom habitats of Western Bank (Scotian Shelf) show that this fishing method leads to reductions in the abundance of sensitive taxa and an overall reduction in epifaunal biomass (Kenchington et al. 2006). A longer term perspective, gained by comparing the diversity of larger species of benthic fauna on the scallop fishing grounds of the Bay of Fundy in 1966–1967 and 1997 (Kenchington et al. 2007) showed that attached, fragile, epifaunal, filter-feeding taxa had been replaced by a combination of motile scavengers, motile filter feeders, and robust, burrowing filter feeders. The authors attributed the changes to the long-term effects of trawling.

Although trawling is known to affect diversity at local scales, there has been little systematic analysis of whether these effects are persistent. Clearly, any fishing with towed bottom gears will lead to collateral impact. On habitats subject to high rates of natural disturbance, where the impacts are smaller and recovery times are faster, society and management authorities have treated the collateral damage as acceptable. With respect to the most sensitive habitats, impacts are not

reversible on management timescales, and society often considers that any impact is unacceptable. Societal pressure and responses of management agencies to policy drivers have, therefore, led to the protection of some of the more sensitive habitats from any fishing activity through area closure and (or) a range of measures intended to reduce impacts.

The first pass of a fishing gear on a previously unfished habitat has a greater impact on biomass than subsequent passes (Kaiser et al. 2002). This has consequences for management, since management policies that maintain a relatively constant "footprint" of fishing through time will lead to a smaller impact on a given habitat than policies that lead to continuous movement and redistribution of effort. Further, management actions that lead to the concentration of a given level of effort in a smaller area type also lead to lower impacts. To minimize fishing impacts on habitats, managers need to minimize the size and variability of the spatial footprint of fishing and ensure that management actions, taken for other reasons, do not lead to unintended extension and (or) displacement of the spatial footprint of fishing. In addition, access to high-resolution habitat maps can allow fishers to operate close to sensitive habitat without unduly disturbing it (Kostylev et al. 2001).

Quantitative estimates of the areas of different habitat types impacted by fishing are not available for Canadian waters, but ongoing efforts to map habitats (e.g., Kostylev et al. 2001) and relatively fine-scale information on the distribution of fishing activities from Vessel Monitoring Systems do make this possible. The analysis could extend to an assessment of total impact, using estimates of the relative impacts of fishing on different habitats types. This would allow managers to assess the impacts and the sustainability of the impacts attributable to different fisheries. However, no benchmarks or reference points have yet been set for acceptable impacts of fishing, and impacts tend to be considered on a case-by-case basis. With, for example, increasingly large areas of glass sponge and cold-water coral reef being identified, the debate on appropriate targets for protection, and whether there is an option for any fishing in areas that harbour such habitats, has not yielded definitive conclusions. Nonetheless, DFO has assessed the potential impacts of fishing gear on benthic habitats, populations, and communities on at least two occasions (DFO 2006, 2010f). The 2006 advice identified the need to establish operational objectives for the conservation of benthic communities and recommended that these objectives should underpin criteria for the establishment of Large Ocean Management Areas (LOMAs; see also Hutchings et al. 2012a). This process was based on the identification of habitats of particular ecological significance, according to DFO criteria for identifying ecologically or biologically significant areas (EBSAs) and ensuring that habitat conservation efforts give priority to those areas.

8.4. Effects on marine fish communities

The effects of fishing on the size and species composition of multispecies communities have been well described, especially in those areas of the continental shelf where annual trawl surveys are conducted. The community effects of fishing are often most profound in areas that are fished with relatively unselective gears, such as bottom trawls, which harvest a mixture of targeted and nontargeted species. Spatial comparisons between areas subject to different fishing intensities, and

temporal comparison in areas where fishing effort is known, indicate broadly consistent community responses.

As fishing mortality rises, the mean size of individuals in the community falls, and species with larger body sizes form a smaller proportion of community biomass. These responses are largely a consequence of three processes: the extent to which higher mortality rates affect larger species, the higher sensitivity of larger species to a given rate of mortality, and the responses of prey species to reductions in their predators. Zwanenburg (2000), for example, showed that the average individual weight of fishes caught in survey trawls on the Scotian Shelf since the 1970s decreased by 51% on the eastern shelf and 41% on the western shelf. Declines in size were greatest for species targeted by commercial fisheries. The slope of the size spectrum (the relationship between the abundance of fishes in body mass classes and body mass) also became steeper, reflecting the relative losses of larger fish and increases in the proportional representation of smaller fishes. After the closure of the cod fishery on the eastern shelf in 1993 and the restrictions on landings on the western shelf both average weights and community size structure stabilized. A subsequent analysis for the western Scotian Shelf by Shackell et al. (2010) showed that decreases in average body sizes had occurred in all functional groups (Fig. 79). Reductions in average mass, relative to values during the 1970s, amounted to 59% for the large-sized benthivores, 48% for medium-sized benthivores, and 45% for piscivores. Planktivores and zoopiscivores declined by 34% and 18%, respectively. For prey species, biomass increased, despite the decline in average body size. The decline in average body size of aggregate top predators was the dominant factor accounting for the increase in prey species biomass. Reductions in abundance of species on the Scotian Shelf and Bay of Fundy led to decreases in the distributional ranges of about half the species studied, an effect likely mediated by changes in interactions within and among species (Fisher and Frank 2004).

The hypothesis that fishing-induced reductions in predator abundance allows for a "release" (abundance increase) of their prey (e.g., Worm and Myers 2003) is further supported by research in the southern Gulf of St. Lawrence. Mean body length in the marine fish community decreased dramatically in the 1990s, a reflection of the removal of large-bodied fishes by fishing and sharp increases in the abundance of small fishes (Benoît and Swain 2008). Strong predator control of the abundance of small-bodied fishes is suggested by the observed inverse correlation between the biomass of small fish and an index of predation on those fish by larger fish (Fig. 80).

8.5. Effects on species community structure and food webs

Most fisheries are managed on the basis of single-species population dynamics models. This means that fisheries reference points (defined in section 5, expanded upon by Hutchings et al. 2012a) are based on how harvesting is predicted to affect the population growth rate of the targeted species of interest, with no consideration as to how changes in the abundance of the targeted species might affect the abundance of interacting species. Reference points that account for these interactions will differ from single-species reference points (Worm et al. 2009).

Fisheries-induced changes to marine species assemblages have been best documented in Atlantic Canada (e.g., Lotze and Milewski 2004), where the overfishing of several species has

Fig. 79. Mean length (*a*) and mean mass (*b*) for five marine fish functional groups from 1970 to 2008 on the Scotian Shelf. Linear regression equations of body size through time are shown. Dots are annual values and lines show 3-year running averages. Grey lines denote the direct measure of growth (size at age 6 year, as weighted by species biomass within each functional group). The *x*-axis represents time from 1970 to 2008 for each of the five functional groups. Source: Shackell et al. (2010).
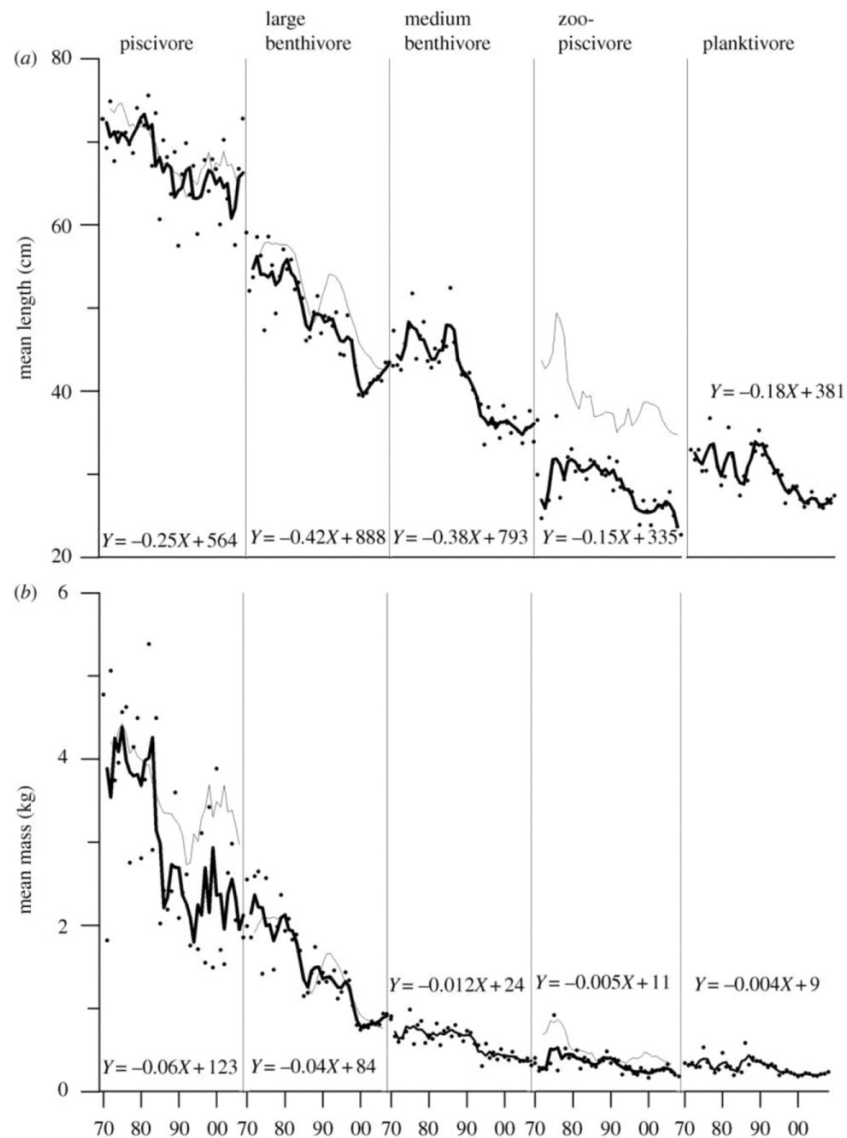
contributed to an unprecedented change in Canadian marine biodiversity (Benoît and Swain 2008; DFO 2010*f*; Templeman 2010; Frank et al. 2011). Perhaps the most demonstrable consequence of these altered ecosystems is an increase in species that were once heavily preyed upon by Atlantic cod and other bottom-dwelling fish predators. One such example is the dramatic increase in shrimp (*Pandalus borealis*) and snow crab (*Chionoecetes opilio*) that followed the collapse of cod in the early 1990s (Worm and Myers 2003).

There are concerns that fishery-induced changes to predator–prey interactions may be responsible for significantly retarding, or even preventing, the recovery of depleted marine fishes. At least three species in the southern Gulf of St. Lawrence are experiencing unsustainably high levels of natural mortality, meaning that they will be extirpated from Canadian waters if mortality rates do not decline. White hake (*Urophycis tenuis*)

in this area may be the most endangered marine fish in Canada. In the 1970s and 1980s, approximately 18% of hake (aged 5–7 years) were dying annually; in the past decade, this annual level of natural mortality has increased to between 86% and 91% (DFO 2010*g*). At this level of mortality, white hake might disappear from the southern Gulf within the next decade. Unsustainably high mortality is predicted to prevent winter skate from increasing, following its 98% decline in the southern Gulf. Atlantic cod in the southern Gulf, once (in 1987) the largest spawning population of cod in the world (Hutchings and Rangeley 2011), are currently experiencing such high mortality that they are projected to be extirpated by 2050 (Swain and Chouinard 2008). One factor originally thought to be inhibiting their recovery is the increase in abundance of species, such as mackerel (*Scomber scombrus*) and Atlantic herring (*Clupea harengus*), that prey upon cod

Fig. 80. Total annual biomass of small fish (dashed line, units on left vertical axis) and an index of potential predation by piscivorous fish on those small-bodied fish (solid line, units on right vertical axis) from 1971 to 2005. Source: Benoît and Swain (2008).
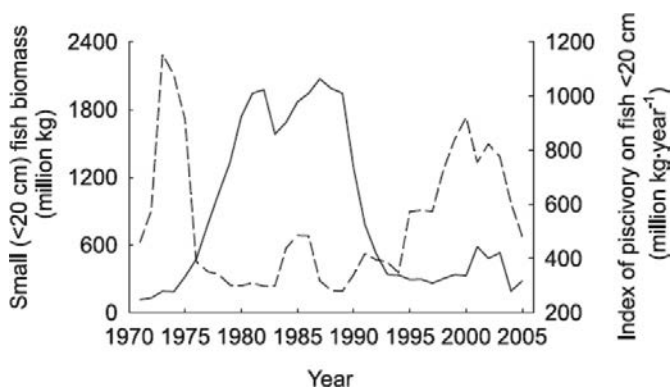
eggs and larvae (and potentially compete with larval and juvenile cod) (Swain and Sinclair 2000). In addition to this hypothesized negative influence on cod productivity is predation on small and large cod by grey seals (DFO 2010g; Benoît et al. 2011; Swain 2011).

Overexploitation on Atlantic Canada's Scotian Shelf has also resulted in the restructuring of a food web. Formerly dominated by large-bodied demersal fishes such as cod and haddock (*Melanogrammus aeglefinus*), the species assemblage is now dominated by small pelagic fishes and macroinvertebrates. It has been posited that recovery of the former species community structure is possible, but that it has been severely delayed by the eruption of planktivorous species (formerly preyed upon by large-bodied predators) and a subsequent outstripping of their zooplankton food supply (Frank et al. 2011).

8.6. Recovery from overfishing

One common means of defining overfishing is to assess current fishing mortality (as defined by a parameter F) with the fishing mortality at which the MSY is estimated to be obtained (i.e., F_{MSY}). When the ratio of F/F_{MSY} exceeds 1, overfishing (as defined by many jurisdictions) is occurring. Based on this metric, one can conclude that overfishing was a dominant characteristic of Canadian fisheries for marine fishes, most notably in the Atlantic, particularly from the 1960s through the early 1990s (Fig. 80).

Reductions in abundance concomitant with overfishing are often associated with significant changes to fish life-history traits, i.e., characteristics that affect the ability of individuals to reproduce and the ability of populations to grow, or recover, following depletion (Hutchings and Baum 2005). Changes to life-history traits affect population growth (Cole 1954) and, thus, recovery potential (Hutchings et al. 2012b). Prominent among fishing-induced changes to life history are reductions in age at maturity, i.e., first reproduction. In at least four populations of Atlantic cod, for example, age at maturity has declined by 2 and 3 years over the past five to six decades (Hutchings and Rangeley 2011). Within some populations, reductions in size at maturity have also been substantive. Length at maturity among Eastern Scotian Shelf cod, for example, declined from ~42 cm in the late 1970s to ~32 cm in the early 2000s (Hutchings 2005).

A second prominent feature of depleted populations is a narrowing of the breadth of ages and body sizes caused by the fishing-out of the largest and oldest individuals. There are theoretical and empirical reasons for believing that population productivity and recovery potential are affected by reductions in the breadth of the age and size classes of the spawning population (Venturelli et al. 2009; Hutchings and Rangeley 2011). All depleted Canadian cod populations have, to varying degrees, experienced significant truncations in their age and size distributions. Newfoundland's northern cod provide an illustrative example. The contribution of eggs by females 10 years and older to the population is estimated to have declined from an annual average of 30% in the 1960s (46% in 1962, when the population had already been fished for almost 500 years) to 17% in the 1970s and to 12% in the 1980s (Hutchings and Myers 1994). Their current low incidence in fishery and survey catches (Brattey et al. 2009) suggests that cod 10 years and older have contributed little to population growth since 1992.

A third feature hypothesized to be of importance to severely depleted populations is genetic change resulting from prolonged periods of overfishing (Stokes et al. 1993; Kuparinen and Merilä 2007; Hutchings and Fraser 2008). The potential for fishing to cause evolutionary change is not appreciably different from other forms of predator-induced mortality, given the ability of fishing to effect differential mortality among genotypes. (The mortality wrought by humans on fishes often exceeds that of most natural predator–prey relationships.) Thus, changes to traits such as age and size at maturity might be explained, in part, as evolutionary responses to unsustainably high levels of fishing mortality. From a recovery perspective, if life-history change partly represents a genetic response to fisheries-induced selection, the reversibility of these life-history responses to exploitation may be slow. Changes to life-history traits and truncated distributions in age and size at maturity have the potential to negatively affect population growth rate and, thus, population recovery as a consequence of several factors, which include higher post-reproductive mortality, reduced lifespan, lower fecundity, smaller egg size, and increased temporal variability in offspring survival (Hutchings 2005; Kuparinen et al. 2012).

The observation that many Canadian marine fishes (e.g., Atlantic cod, American plaice, winter skate, white hake) have shown little or no recovery following depletion, despite massive reductions in harvesting, indicates that reductions in fishing mortality, while being necessary for recovery, are not always sufficient to achieve it. However, given the necessity for fishing pressure to be reduced before recovery can take place, the declining trend in fishing mortality across all fish populations for which data are available in Canadian waters is encouraging (Fig. 81).

9. Aquaculture: observed and projected consequences for Canadian marine biodiversity

9.1. Introduction

The growth and monetary value of Canadian aquaculture (section 6) mask real and perceived environmental costs and a deeply rooted public controversy. Potential environmental impacts of aquaculture, which might have biodiversity conse-

Fig. 81. Temporal trend in fishing mortality (F, the instantaneous mortality rate) relative to the fishing mortality estimated to achieve maximum sustainable yield (F_{MSY}) (blue line) for commercially exploited fish stocks in the Canadian Pacific and Atlantic. Number of fish stocks for which data are available in each year is shown by the black line. Note that when F/F_{MSY} exceeds 1, overfishing (as defined by many jurisdictions) is occurring. Sources: Table 3 and RAM legacy 2012.
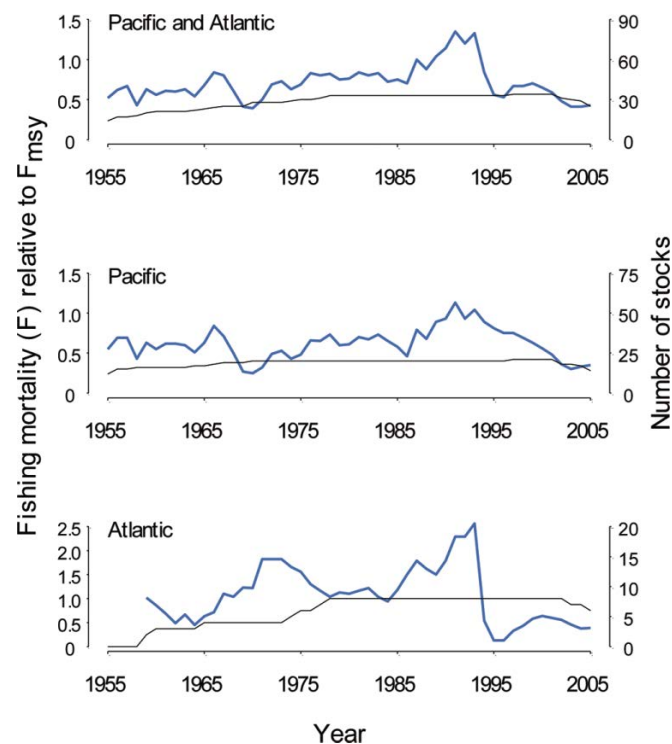
quences, are commonly grouped into four categories: ecological interactions, genetic consequences, disease and parasites, and habitat alteration (Fig. 82). More specifically, these include concerns about (*i*) benthic impacts and siting; (*ii*) chemical inputs, such as antibiotics, antifoulants, and pesticides; (*iii*) nutrient loading and deterioration of the benthos; (*iv*) attraction of other organisms and predator exclusion; (*v*) feed sources; (*vi*) effects of escapees and use of exotic species; and (*vii*) exchange of pathogens, such as sea lice, between the local natural and culture environments (the term "sea lice" refers to naturally occurring species of small, marine copepods in the family Caligidae; these parasitic organisms feed on the mucous, skin, and blood of their hosts). All of these interactions are known to occur in the open-net sea pens that are typical of Canadian aquaculture operations (it should be noted, however, that most or all of these interactions can be mitigated by the use of closed-containment facilities, particularly those deployed on land). The extent of their impact varies significantly depending on the species involved, spatial location of the culture activity (siting), magnitude or scale and type of activity, and local environment.

Unfortunately, background conditions are seldom considered to their full extent, a deficiency that becomes particularly important if we anticipate conditions changing in the future as a result of climate change. Annually varying environmental

conditions can directly influence the assessment of potential impacts. For example, baseline monitoring (predevelopment) and determination of reference farm sites are vital to the assessment of potential farming impacts; the susceptibility of juvenile Pacific salmon to pathogens from aquaculture farms could differ between years because of variability in the natural environment to which they were exposed prior to encountering farms. Much of the public controversy associated with aquaculture stems from the uncertainty of their impacts on natural systems (impacts can be localized, transient in time, and differ between sites), lack of an assessment framework for analysis (in the absence of a monitoring and assessment framework, how should impacts be assessed?), and reliance on statistical relationships and mathematical models to compensate for this uncertainty and lack of assessment framework. Although modelling is essential to most studies in natural biological systems, models can introduce concerns about understanding and trust in public debate of controversial topics.

The issues outlined above become more complicated when considering biodiversity. Impacts associated with an aquaculture site on local habitats and in a limited area are difficult to extrapolate to biodiversity impacts on a broader scale, or on the production of local native species. Also, effects on biodiversity are more likely to be associated with cumulative effects of culture programmes, and not on any single issue or single culture site. Cumulative effects of culture activities, however, can be difficult to quantify and isolate from other sources of variation in open natural ecosystems. In BC's Broughton Archipelago, for example, aboriginal people have practiced a form of shellfish culture for centuries (Williams 2006). Once natural shellfish beaches, these "clam gardens" were modified to increase clam production. Recently, however, many of the gardens have become less productive and the clams are inedible. Aboriginal people attribute this to salmon aquaculture; 28 farm sites have been developed in the archipelago since the mid-1980s. Questions persist about the role of aquaculture versus changes in how the gardens are used, or changes in local environmental conditions; and research on the issue is incomplete at present.

Globally, published comments on the effect of aquaculture on biodiversity are generally negative and have fueled public concern (e.g., Naylor et al. 2005, 2009; Alder et al. 2008; Ford and Myers 2008; Tacon and Metian 2008; Costello 2009; Burridge et al. 2010; but see Cubitt et al. 2009). (Although these reviews pertain primarily to finfish, Dumbauld et al.'s (2009) recent review of US shellfish aquaculture is relevant in the Canadian context.) The focus here will be on recent work related to biodiversity consequences. In this regard, DFO prepared a number of "State of Knowledge" reports to provide a scientific review of potential environmental effects of aquaculture (DFO 2003–2006; DFO 2011c). Comprehensive as these reports are, their references have become increasingly dated (the most recent citations range typically between 2002 and 2004). The DFO (2003–2006) review includes marine (and freshwater) finfish and shellfish aquaculture and addresses concerns pertaining to impacts of (*i*) waste deposition (including nutrient and organic matter); (*ii*) use of chemicals (e.g., pesticides, drugs, antifouling agents); and (*iii*) interactions–interbreeding between farmed and wild species (including disease transfer and genetic and ecological effects). Among recent DFO reports on aquaculture that have been published on DFO's Canadian Science Advisory Secretariat Web site (DFO

Fig. 82. Environmental impacts associated with open-net pen aquaculture. Impacts associated with other forms of open-ocean aquaculture will differ but commonly involve this same set of potential effects. Source: Ocean Conservancy (2012).
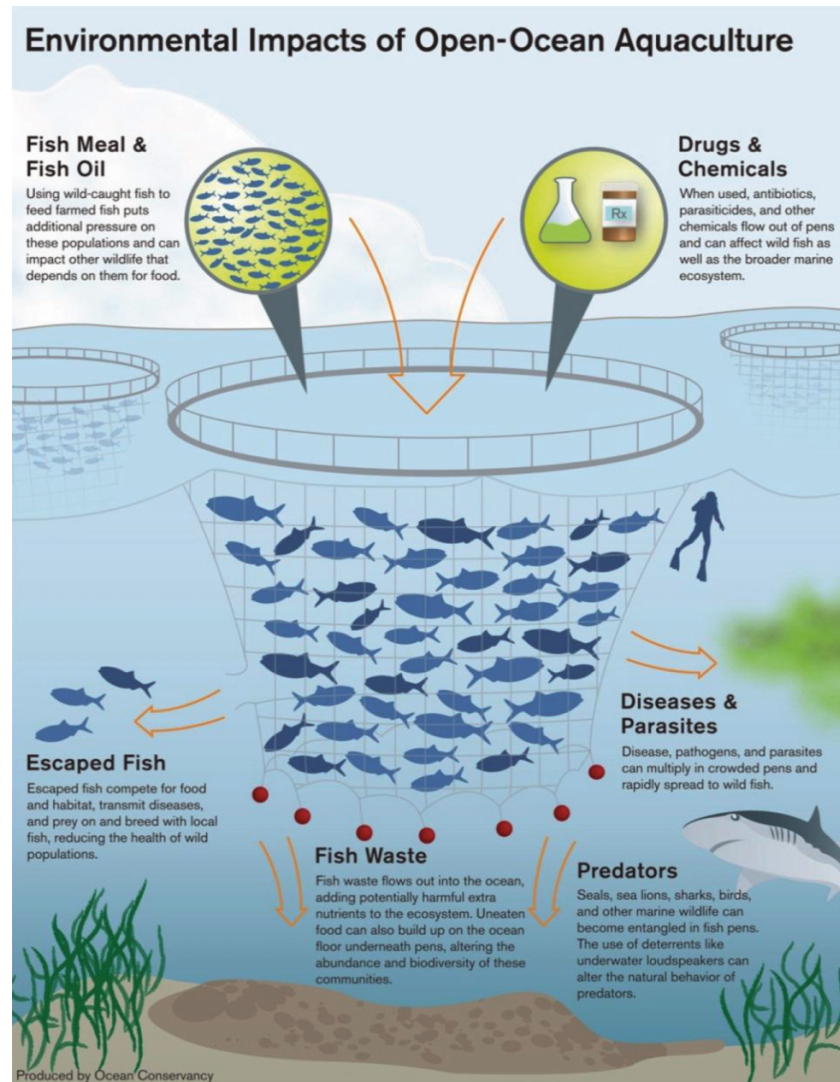
2010*h*) is a study of the degree to which aquaculture-related alterations to physical habitat acts as an ecosystem stressor (McKindsey 2010).

9.2. Finfish aquaculture

9.2.1. Pacific

With the possible exceptions of pathogens, it is unlikely that the impacts of salmon net-pen aquaculture on marine biodiversity along BC's coast will be broad-ranging. Effects, however, are likely to be cumulative, particularly in areas of salmon farm concentration. Concerns about escaped Atlantic salmon, an exotic species in BC, were significant in the 1990s but have decreased in recent years. Escapes of salmon certainly occur but are less frequent than they were in the developmental years of the aquaculture industry. DFO reports that, between 1987 and 2002, 1.4 million Atlantic salmon escaped from BC salmon farms. In more recent years, escapes and catches of Atlantic salmon have been reported by the Aquatic Nuisance Species Project in Portland, Oregon (Aquatic Nui-

sance Project 2006) (Table 5). Although escapes of salmon from BC farms must be reported to regulatory agencies, the number of escapees is clearly an estimate, and the recovery of Atlantic salmon depends on monitoring and sampling efforts, both of which vary considerably among years. (For example, DFO's Atlantic Salmon Watch programme has ceased.) Recoveries in Alaskan fisheries may be the best indicator of incidence of Atlantic salmon in Pacific coastal waters, due to the consistency of their fisheries and sampling programmes.

Although feral Atlantic salmon have been reported in coastal rivers (Volpe et al. 2000), there is no evidence, to the best of our knowledge, of self-reproducing populations of feral Atlantic salmon in BC. If a concern for escapees exists, it is more likely in Clayoquot Sound (west coast of Vancouver Island), where domesticated strains of Chinook salmon are reared in proximity to native Chinook populations (Kim et al. 2004; Withler et al. 2007). These native populations are also depressed in abundance and potentially vulnerable but, to our knowledge, monitoring programmes to assess the potential

Table 5. Reported escapes of Atlantic salmon from BC salmon farms and their recorded catches in Canadian and Alaskan fisheries.

Year	Reported losses	Canadian catches of escapees	Alaskan fisheries
2000	31 855	7834 (131)	81
2001	55 414	179 (116)	35
2002	11 257	562 (40)	6
2003	30	46 (36)	3
2004	43 969	148 (0)	1
2005	21	27 (2)	3
2006	17	225 (1)	1
2007	19 223	21 (5)	3
2008	111 679	No data	39
2009	48 857	No data	2
2010	0	Unreported	9

Note: Values in brackets for Canadian catches of escapees are reported from freshwater sampling programmes (i.e., potential spawning fish).

impacts of Chinook salmon aquaculture on native Chinook populations in Clayoquot Sound are not conducted.

The greatest concerns for biodiversity impacts are associated with the effects of sea lice from salmon farms on local populations of wild Pacific salmon, the potential for exchange of pathogens between farmed and wild salmonids, and the risk of introducing new pathogens to the wild. Although salmon farming in BC was initiated in the mid-1980s (section 6), public and scientific focus on sea lice was not strong until the reporting of an epidemic in the Broughton Archipelago in June 2001 (Morton and Williams 2003). While essentially all adult salmonids returning from the Pacific Ocean carry sea lice naturally, juvenile salmonids in coastal waters do not. Observations of wild juvenile salmonids infected with lice in the vicinity of farms raised a series of questions: What is the source of the lice on juvenile salmonids? What is the effect of lice infections on individual juveniles entering coastal waters? Is there an effect on the productivity (the rate of adult return per adult spawner in the parental generation) of salmon populations in proximity to salmon farming?

After a decade of study, it is generally accepted that open-net pen salmon farms can cause infections of the salmon louse (a type of sea lice), *Lepeophtheirus salmonis,* and contribute to infections of *Caligus clemensi* in native salmonids, and these infections can increase juvenile salmonid mortality rates (directly and probably indirectly through increased predation). Although it has also been hypothesized that increased juvenile mortality might reduce the productivity of salmon populations (e.g., Saksida et al. 2007a, 2007b; Beamish et al. 2009; Marty et al. 2010; Krkošek and Hilborn 2011; Krkošek et al. 2011a, 2011b; Price et al. 2010, 2011), definitive links between juvenile infections and adult salmon abundance have been difficult to establish. The number of adults returning to a specific region reflects variation in adult spawning numbers in the preceding parental generation, and conditions experienced subsequently by their progeny. This usually involves returns to multiple streams, variable environmental conditions in fresh water, and naturally variable conditions in the ocean between the juvenile and returning-adult stages. The natural mortality

rate on all salmonids entering the ocean is known to be high (usually more than 90% die), so the ability to assess an incremental impact of sea lice during an early life stage will always be limited, particularly given the quality of data typically available for analysis.

Data obtained from the Broughton Archipelago demonstrate that sea lice infections on wild salmonids can be controlled through treatment of farmed salmon prior to the entrance of juvenile salmon into coastal waters (Saksida et al. 2010). Current treatment in BC involves the use of emamectin benzoate (SLICE®, Schering-Plough Animal Health, Olds, AB) as a premixed coating applied to fish food pellets and administered under veterinary supervision. For government-approved chemicals, recent research in Canada indicates limited impacts at the dosages currently applied (Waddy et al. 2007; Kuo et al. 2010) and that levels retained in flesh and skin are well below Health Canada guidelines (Whyte et al. 2011). (However, a law suit filed by Environment Canada against Cooke Aquaculture in autumn 2011 alleges that cypermethrin, an illegal pesticide, has been used to combat sea lice infestations in Atlantic salmon farms in the Bay of Fundy, leading to deaths of lobsters.) One emerging concern is the development of sea lice resistance to emamectin benzoate and reduced effectiveness in controlling lice infections. While variation in up-take of SLICE between animals has been documented (Berg and Horsberg 2009), the question of whether sea lice resistance is developing in farmed fish remains unanswered and merits study. The control of sea lice is also important for the control of infectious diseases (Nese and Enger 1993; Nylund et al. 1994; Baker et al. 2009).

The introduction or exacerbation of infectious diseases and parasites probably constitutes the greatest potential threat to biodiversity posed by salmon aquaculture. Significant disease outbreaks in BC salmon farms have been documented. In one instance, the disease was an endemic one (infectious haematopoietic necrosis virus, IHNV) and it was likely spread within, and between, coastal areas as a result of farming practices (Saksida 2006). To date, diseases in BC salmon farms and government salmon hatcheries have all been introduced by native bacteria or viruses known to infect wild Pacific salmon. However, the recent outbreaks of infectious salmon anemia (ISA) in Chile (Vike et al. 2009), coupled with various disease transmission studies of ISA (e.g., Nylund et al. 2007), have greatly heightened awareness and concern in BC. Indeed, in 2011, there were conflicting reports that wild BC salmon had, for the first time, been infected with ISA. One set of analyses, undertaken by a World Animal Health Organization-endorsed lab at the University of Prince Edward Island, was reported to have found positive evidence of the virus, whereas a later set of analyses on different tissues (albeit apparently degraded) from the same fish by DFO were reported to be negative, according to the Canadian Food Inspection Agency (CFIA 2011). Prior to 2011, all records of disease incidence on BC salmon farms had been protected by the Province of BC as being proprietary to the industry. Although the Cohen Commission into Fraser sockeye salmon is now in receipt of some portion of these records (www.cohencommission.ca; accessed 19–11–2011), it is not yet known how accessible these records will be to the public. The introduction of a nonendemic disease, such as ISA, into BC's wild Pacific salmon would certainly constitute a major threat to the diver-

sity of these species. Miller et al.'s (2011) functional genomics study has contributed strongly to these concerns; the researchers hypothesized that elevated mortality in Fraser sockeye salmon in recent years represents a response to an unknown viral infection (linkages cited with genes associated with leukemia). Clearly, this hypothesis warrants further study.

In summary, while adverse impacts of open-net pen salmon farming on the Pacific coast have been identified, the impact on marine biodiversity in these waters is likely to be localized and controllable, with the exception of potential impacts arising from the transfer of pathogens and disease to wild populations. There is reason to believe that the harm posed by pathogens might be greater than currently perceived. The lack of transparency in public reporting (cf. Principle 10 of the Rio Declaration in Hutchings et al. 2012a) of diseases at aquaculture farms has hindered meaningful, constructive, and respectful debate. Public concern for salmon aquaculture in BC is frequently deflected to scientific arguments and burden of proof. But this type of "objective" approach misses an essential subjective viewpoint (David Suzuki Foundation 2010):

> [Pacific salmon] are a symbol of place in the northwest, a marker of the community of individuals, enterprises and organizations committed to live in a way that strengthens local and regional economies, sustains the natural abundance of resources, and provides a nurturing for the spirit.

The sustainability of salmon farming in BC involves more than science and models and will continue to be debated until there is a fuller understanding and more meaningful inclusion of public values and opinions within aquaculture management and government policy decisions. There are no other regions of the world where open-net pen salmon farming is practiced that have greater salmonid diversity, abundance, and dependent natural ecosystems that are potentially at risk than those in BC. The use of closed-containment technology has been proposed as a solution to mitigate some of the impacts of open-net pen salmon farming (EPI 2008) and land-based closed-containment fish farming may now be economically viable (e.g., Wright and Arianpoo 2010; Sustainable Blue 2012). Ultimately, a higher standard of transparency and accountability by both industry and DFO should have been anticipated but has yet to be achieved.

9.2.2. Atlantic

Given that the farming of salmon dominates finfish aquaculture on both coasts (section 6), the projected consequences of the industry on marine biodiversity in the Atlantic are in many ways similar to those in the Pacific. Here, the Report focuses on aspects of escapes and disease issues that are somewhat more specific to the Atlantic region.

Unlike the Pacific, Atlantic salmon are native to Atlantic waters and the potential for negative impacts due to intraspecific competition and genetic introgression — the infusion of genes from escaped farmed fish into wild fish — are manifest (Fleming et al. 2000; McGinnity et al. 2003; Fraser et al. 2010). The scale of the industry, relative to the small size of wild salmon populations in the region, increases the likelihood of ecological and genetic impacts, even at relatively low escape rates (cf. Hindar et al. 2006). Concerns are further accentuated in the Atlantic because the industry is concentrated in areas where the abundance of wild salmon populations continues to be depressed (e.g., Bay of Fundy, south

Fig. 83. Yearly mean proportions of escaped farmed Atlantic salmon within spawning runs, averaged over all eastern North American rivers investigated. Peaks appear to correspond with large episodic escape events, which are indicated by arrows with the estimated number of escapees noted. Source: Morris et al. (2008).
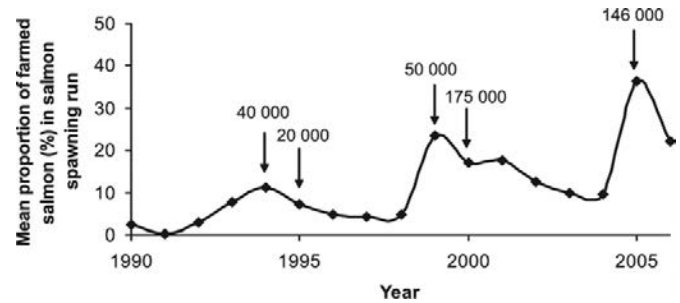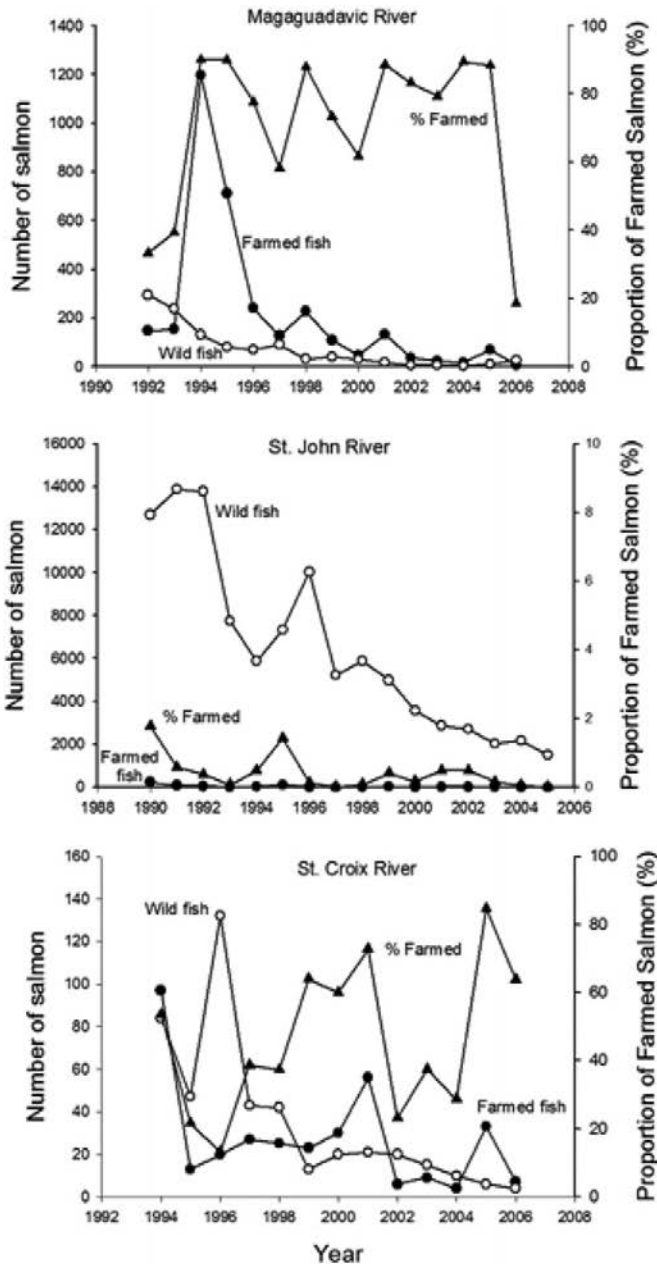

coast of Newfoundland; DFO and MRNF 2008, 2009). At a regional scale, the proportion of farmed salmon invading wild populations has increased through time (Fig. 83), although this trend appears to be more of an effect of declines in wild populations than increases in the absolute number of farm escapees (Fig. 84). Escaped farmed salmon have been reported in 54 rivers and bays, which constitute 87% of the watersheds that have been investigated since the inception of the salmon aquaculture industry (Morris et al. 2008).

There are important differences between farmed and wild salmon, such as growth rate, that affect behaviour, competitive ability, and breeding success (reviewed by Thorstad et al. 2008). These characteristics are caused partly by environmental differences and partly by genetic differences and include responses to intentional and unintentional selection (domestication) in aquaculture facilities. While the outcomes of interactions between farmed and wild salmon depend on context, varying with a number of environmental and genetic factors, they will frequently be negative for wild salmon. Genetic introgression resulting from the interbreeding of farmed and wild salmon can disrupt adaptive traits (McGinnity et al. 2003; Fraser et al. 2010) and lead to genetic homogenization (i.e., a diminishment of between-population distinctiveness), which might further threaten the persistence of wild populations (Hindar et al. 2006).

Regarding pathogens of major concern in Atlantic Canada, the ISA virus has already caused enormous economic losses to salmon aquaculture and constitutes a threat to wild salmon populations because of the magnification of pathogen abundance within open-sea net cages. Surveillance of farms in the Bay of Fundy, subsequent to the first report of ISA in 1997, has identified 20 genetically distinct ISA virus isolates of differing levels of virulence (Leadbeater and Glebe 2010). Considerable efforts have since been expended to develop effective vaccines and establish biosecurity plans to control the spread of disease when outbreaks occur. None of the current methods, however, provides a complete barrier to disease transfer between farmed and wild fish (Hammel et al. 2009); the risk of pathogens spreading to wild populations persists.

Many of the general environmental concerns associated with salmon aquaculture apply to the aquaculture of other marine fish raised in open-containment facilities, such as steel-

Fig. 84. Numbers of farmed and wild Atlantic salmon over time in three river systems in New Brunswick. Proportions of farmed salmon have varied temporally but have remained consistently high in some cases, despite declines in the numbers of farmed escapees present. This largely reflects declines in wild populations. Source: Morris et al. (2008).

demand for seafood and subsequent industry growth. Moreover, as the aquaculture of different species of marine finfish share many of the same properties (e.g., net cages sited in the ocean), many of the concerns will be cumulative, regardless of the species of finfish being reared. It is also possible that warming temperatures associated with climate change will affect aquaculture production by increasing the number of potential sites, the growth rates of farmed species, and the numbers of species amenable to farming, but projections of this nature have not been undertaken.

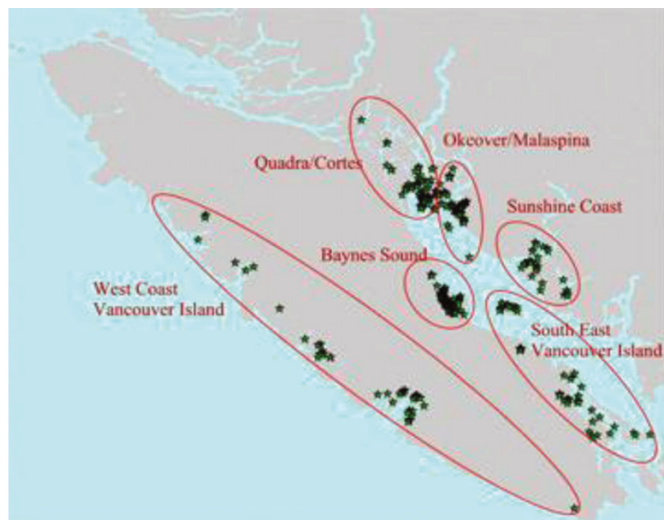## 9.3. Shellfish aquaculture

9.3.1. Pacific

Shellfish culture is the oldest sector of the aquaculture industry on the west coast of North America. Pacific oyster (*Crassostrea gigas*) was first introduced to BC from the Far East in 1912, although most seed was imported from Japan between 1930 and 1939. The species that dominates clam farming in BC (Manila clams, *Venerupis philippinarum*) was inadvertently introduced during importation of Pacific oyster seed. Manila clams were first observed in the natural environment in 1936 and are now distributed through central BC. Japanese scallop (*Patinopecten yessoensis*) was also introduced and is now the most important cultured species both in production and value. Other species that are commercially produced include the European oyster (*Ostrea edulis*), the littleneck clam (*Protothaca staminea*), the Pacific scallop (a cultured hybrid; Bourne and Bunting 2009), blue mussel, Gallo mussel (*Mytilus galloprovincialis*), and the geoduck clam (*Panope abrupta*). Marine species being considered or under early development for culture include northern abalone, sea cucumber (*Parastichopus californicus*), green sea urchin (*Strongylocentrotus droebachiensis*), and cockles (*Clinocardium nattallii*). For successful commercial operations, hatchery culture and rearing of juveniles has been developed for most of these species. Additional information is available from an informative web site supported by the BC Shellfish Growers Association (BCSGA 2012).

The potential environmental consequences of shellfish aquaculture are similar to those associated with finfish. Ecological concerns include (*i*) changes to shellfish diversity within tenure areas caused by species removal for farming purposes; (*ii*) altered use of the tenure area by other species, including fishes (McKindsey et al. 2006, 2011; McKindsey 2010) and birds (Booth 2001); and (*iii*) cumulative effects, particularly in areas of dense utilization such as Baynes Sound, an area used by at least 12 species of seabirds of global and continental concern (Booth 2001) (although positive associations between the densities of surf scoters (*Melanitta perspicillata*) and white-winged scoters (*M. deglandi*) and farmed mussel densities have been reported; Zydelis et al. 2009). Eighty to 90% of the shoreline of Baynes Sound is under shellfish tenure, and Wan and Bendell-Young (2010) estimated that 56% of what they term "viable intertidal habitat" is under culture.

Genetic-based concerns include those resulting from species interactions (three exotic species noted above) reported for mussels (Shields et al. 2008). The introduced Japanese oyster has also contributed to heightened conservation concern for the endemic Olympia oyster (Gillespie 2009) and its recovery (Trimble et al. 2009). Diseases and parasites, prevalent in any

head, Atlantic cod, Atlantic halibut (*Hippoglossoides hippoglossus*), spotted wolfish (*Anarhichas minor*), and Atlantic wolfish (*A. lupus*). An additional concern may manifest itself in regard to the containment of fishes, such as cod, capable of spawning within net cages and releasing viable embryos into the wild (Jørstad et al. 2008). However, the aquaculture of nonsalmonid fish in the Atlantic is currently of a sufficiently small scale that the individual impacts of their operations are unlikely to be considerable. This could change with increased

Fig. 85. Distribution of 458 shellfish tenures in southern British Columbia, identified by geographic clusters (two tenures exist in northern BC). Source: Vancouver Island University, Centre for Shellfish Research Web site (http://www.viu.ca/csr/industry/; accessed 5–8–2011).

Table 6. Main species of seaweed grown in BC's aquaculture industry. Source: www.dfo-mpo.gc.ca/aquaculture/marine-eng.htm; accessed 19–11–2011.

Common name	Scientific name
Alaria	*Alariamarginata* — Winged Kelp
Brown Algae	*Costariacostata* — Five Rib Kelp
Green Algae	*Enteromorpha* spp. — Green String Lettuce
Red Algae	*Gelidium* — Gel Weed
Kombu	*Laminaria saccharina* — Tangle
Groenlandica	*Laminaria groenlandica* — Tangle
Giant Kelp	*Macrocystis integrifolia*
Bull Kelp	*Nereocystis luetkeanna*
Nori	*Porphyra* spp. — Purple Laver
Sea Lettuce	*Ulva lactica*

form of intensive animal culture, have been significant during industry development (Bower and McGladdery 2003) but are apparently under control, based on the observation that the BC Shellfish Growers Association does not identify them as a research and development priority. One known introduction of a parasite, previously unreported in BC, has been associated with shellfish aquaculture (Marty et al. 2006).

Habitat alterations are extensive within tenures, particularly in the benthos (Bendell-Young 2006; Whiteley and Bendell-Young 2007; but also see Munroe and McKinley 2007) and associated shorelines (Bendell-Young et al. 2010). Use of chemicals is apparently limited, but effects on aquatic primary production can be expected, considering the density of tenures in southern BC. However, these effects will be localized and temporary. Similar issues were previously addressed by Deal (2005) with respect to the development of a sustainable shellfish aquaculture industry in BC. Invasive species have only been noted when attributed to activities related to shellfish aquaculture, yet other species may become problematic for the industry or could affect the natural diversity of BC's coastal waters. One such example is the purple varnish clam (or savory clam, *Nuttallia obscurata*), which entered the Strait of Georgia via ballast water in the later 1980s and is now widely distributed (Dudas and Dower 2006).

Dating back to the 1930s, and entailing the introduction of its major utilized species, BC's shellfish aquaculture industry has probably had limited effects on marine biodiversity, especially when considered on a coast-wide scale. Some individuals have even suggested that positive benefits accrue to natural ecosystems in the guise of improved water quality and the increased productivity of some native species. Nonetheless, the distribution and growth of the industry suggest that ecological impacts could be substantial in specific areas (Fig. 85), although potential consequences to biodiversity have not yet been specifically assessed.

9.3.2. Atlantic

In contrast to the Pacific, shellfish aquaculture in the Atlantic involves predominantly native species (blue mussel; horse mussel, *M. trossulus*; eastern oyster, *Crassostrea virginica*; sea scallop, *Placopecten magellanicus*; softshell clam, *Mya arenaria*; and the hard clam or quahaug *Mercenaria mercenaria*) (section 6). Exceptions to this include small-scale cultivation of bay scallop (*Argopecten irradians*) from the US Atlantic coast, Iceland scallop (*Chlamys islandica*) and European oyster (*Ostrea edulis*) from the Northeast Atlantic, and red abalone (*Haliotis rufescens*) from the US Pacific coast. Thus, the potential exists for the establishment of nonnative species in the wild and associated negative ecosystem effects resulting from such invasions (Ruesink et al. 2005; McKindsey et al. 2007; Forrest et al. 2009). The industry itself is threatened in parts of Atlantic Canada by noncultured invasive species, including the vase tunicate (*Ciona intestinalis*) and green crab (*Carcinus maenas*) from the Northeast Atlantic; oyster drill (*Urosalpinx cinerea*) from the northwest Atlantic; the Mediterranean golden star tunicate (*Botryllus scholesseri*); and clubbed tunicate (*Styela clava*), violate tunicate (*Botrylloides violaceus*), the alga *Codium fragile tomentosoides*, mitten crab (*Eriocheir sinensis*), and the tunicate (*Didemnum vexillum*) from the western Pacific.

Many of the other concerns outlined in relation to shellfish culture in Pacific Canada apply to Atlantic Canada, including impacts on benthos, collection of wild spat–seed, transfer of diseases and other organisms associated with movement of shellfish and seed, predator attraction and exclusion, conflict with capture fisheries over access to coastal fishing grounds, chemical inputs, and local changes to primary productivity. An integrated multitrophic aquaculture (IMTA; CAIA 2012) approach that combines finfish, shellfish, and algae culture may ameliorate some impacts and reduce the ecological footprint.

9.4. Seaweed aquaculture

At present, there are 10 species of seaweed that constitute the primary aquaculture species grown in BC (Table 6). Each of these species is native to BC waters (Druehl 2000) and no invasive species have been recorded (Williams and Smith 2007). Beyond a simple species listing, there seems to be extremely little documentation on marine plant culture in Canadian Pacific waters.

The main seaweeds cultured in Atlantic Canada are the native Irish moss (*Chondrus crispus*) and knotted wrack (*Ascophyllum nodosom*). Irish moss has been harvested commercially in Atlantic Canada since at least 1948 and is used as an industrial source of carrageenan, which serves as a thickener and stabilizer in milk products and processed foods. Knotted wrack is grown for the extraction of alginate, which is used for the production of gels, as a constituent of fertilizers and a gelling or thickening agent. Both species are harvested primarily by drag raking of designated seaweed beds in nearshore and intertidal areas. This technique disrupts near-shore environments and, although the impacts tend to be localized, the harvest does affect nursery areas for juvenile fish and the habitat of other nearshore organisms. As a result, management plans (including quotas, cutting height restrictions, gear restrictions, closed areas) have been implemented to help mitigate these impacts (Ugarte 2007).

10. Summary of key findings

Here we briefly summarize the expert panel's key findings as they relate to trends in three factors affecting Canada's ocean biological life (climate change, fisheries, and aquaculture) and to the realized and potential consequences that each has on marine biodiversity. The full set of citations that support these key findings can be found in the preceding text.

10.1. Indicators of climate change in Canada's oceans

Canada's coastline and ocean surface area are greater than those of most countries. Surface water temperatures are increasing and high-latitude waters are becoming less salty. This warming and freshening of the oceans can reduce the transport of nutrients from deep waters to surface waters. A nearly ice-free Arctic summer could occur as early as the late 2030s. Increasing sea levels are forecast to lead to increased flooding, coastal erosion, and saltwater intrusion into wetlands and ground water. Canada's oceans are also becoming increasingly acidic, and oxygen levels have been declining; in some areas, oxygen levels are so low that affected waters are now unsuitable for most aquatic life.

10.2. Trends in Canadian marine biodiversity

More than 16 000 marine species have been recorded in Canada, although there may be at least 2–3 times as many species still to be found. The Pacific is particularly rich in seaweed species; the Arctic, in small crustaceans; and the Atlantic, in fishes; Canada hosts 40% of the world's marine mammal species. Any increase in the number of marine species assessed as being at risk in Canada will likely be attributed to forthcoming assessments of Pacific salmon populations and population groups. Species assemblages of plankton are sensitive to changes in water temperature which, in turn, affect the quantity and quality of food available to invertebrates and fishes. Marine fishes in Canada's oceans are estimated to have declined in abundance by an average of 52% from 1970 to the mid-1990s and have remained stable thereafter; most commercially fished stocks remain well below conservation target levels. Most, but not all, marine mammals have increased following past over-exploitation. Trends in seabirds have been mixed, showing increases in some areas and declines in others.

10.3. Trends in Canadian marine fisheries and aquaculture

In 2009, Canada's fishery catches were half those of the late 1980s; the landed value of all fisheries in 2009 was almost the lowest since 1977. Atlantic fisheries, once predominantly for bottom-dwelling fishes, are now dominated by lobster, shrimp, and crab; Pacific catches have experienced marked declines in salmon. Marine aquaculture, dominated by the farming of Atlantic salmon, experienced rapid growth from the early 1980s until 2002; production has since stabilized. British Columbia is the 4th largest producer of farmed salmon in the world and farms 67% of Canada's finfish aquaculture. Shellfish production, having grown considerably since the 1980s (valued at $736 million in 2009), is dominated by Atlantic Canada's culture of blue mussels.

10.4. Climate change: consequences for Canadian marine biodiversity

Climate change affects the physiology, development, reproduction, behaviour, food supply, and survival of marine species by influencing factors such as water temperature, salinity, oxygen, and acidity. Species are projected to shift their latitudinal and depth ranges, changing the community composition of native marine species and allowing for invasions of nonnative species. Climate change is acting to decouple the timing of resource requirements and resource availability for some species, impairing their reproduction and development. The effects of ocean acidification on marine biodiversity, although not yet well understood, are likely to be far-reaching and complex.

10.5. Fisheries: consequences for Canadian marine biodiversity

Fishing affects biodiversity primarily by reducing abundance, sometimes significantly, as a result of directed catches, bycatch, and the destruction of species or their habitat (e.g., corals and sponges). Overfishing has depleted many fish stocks, potentially increasing their chance of extinction. By affecting abundance, fishing alters interactions among species, such as those between predator and prey, resulting in dramatic changes to marine ecosystems and food webs. Fishing mortality of marine fishes has declined since its peak in the late 1980s and early 1990s, although reductions in fishing pressure are not always sufficient to enable recovery.

10.6. Aquaculture: consequences for Canadian marine biodiversity

Bottom-dwelling organisms and their habitat can be affected by organic wastes and chemical inputs, such as antibiotics, antifoulants, and pesticides, in open-sea net-pen facilities. Exchange of pathogens between farmed and wild species can seriously threaten wild species. Interbreeding between wild fish and escapees of the same species threatens the reproductive capability and recovery potential of wild populations of conservation concern. The primary biodiversity concern associated with shellfish aquaculture is the farming of nonnative species.

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