CHAPTER 6: **BIODIVERSITY AND PROTECTED AREAS**

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KEY FINDINGS

Biodiversity – the variety of species and ecosystems and the ecological processes of which they are a part – has a paramount influence on Canada's natural capital and its ability to deliver services. Those services, in turn, contribute to human health and well-being and support a wide range of economic sectors. Climate change, interacting with other human stressors such as pollution and landscape fragmentation, is already impacting biodiversity in Canada. Future impacts will be affected by the magnitude of continuing climate change and adaptation decisions made to enhance ecosystem resilience. Key findings of this chapter include:

- Climate-related shifts in species distributions have already been documented in Canada. Future range shifts will include expansion, contraction and fragmentation in species-specific patterns. In many locations, differential range shifts among species are likely to result in novel ecosystems.
- Phenological mismatches occur when shifts in the timing of life cycle events differ between dependent species and, as an example, can result in migrating species arriving at a site after the peak prey availability has passed. Phenological mismatches are expected to become more frequent in the future, as is hybridization. The impacts of hybridization can drive rare species to extinction or, in other cases, increase the adaptability of some species by introducing genetic variation.
- For many species, the current and projected rates of environmental change are likely to exceed their natural ability to adapt, increasing stress and threatening biodiversity. As a result, climate change is magnifying the importance of managing ecosystems in a manner that enhances resilience and preserves biodiversity.
- Protected areas, including parks, wildlife reserves and marine protected areas will play an important role in the conservation of biodiversity in a period of rapid change. Many protected areas will provide "refuge" or migration corridors for native species, serving to maintain genetic diversity. Protected areas tend to be more resilient than the intervening landscapes and waterscapes because they contain relatively intact ecosystems and are less impacted by non-climate stressors such as habitat loss and fragmentation.
- Many Canadian jurisdictions are expanding their system of parks and other protected areas as part of their overall management plans and climate change adaptation strategies. Initiatives aimed at maintaining or restoring landscape connectivity serve to increase ecosystem resilience by enhancing the capacity for species to adjust their distribution in response to climate change. Associated research, monitoring, citizen science, public awareness, and visitor experience programs build understanding and help to engage the public in meaningful participatory decision-making.
- The conservation community recognizes the value of ecological restoration in strengthening the resilience of ecosystems to climate change. Integration of climate change adaptation strategies into restoration decision-making in Canada, as elsewhere, is complex.

1. INTRODUCTION

Canada is home to major portions of the world's polar regions and tundra, boreal and temperate forests, grasslands and aquatic ecosystems, including the Great Lakes and territorial waters in the Pacific, Arctic and Atlantic oceans. These ecosystems contain about 10% of the world's forests and 20% of the world's freshwater (Environment Canada, 2009). They also provide niche space for more than 70 000 species of mammals, birds, reptiles, amphibians, fish, invertebrates, plants and other organisms (CESC, 2011).

Biodiversity, the variety of species and ecosystems and the ecological processes of which they are a part, is the natural capital that provides the foundation for much of Canada's economic and social well-being. It contributes to cleaner air and water, climate regulation, carbon storage, pollination and flood regulation. Humans directly and indirectly benefit from biodiversity as, for example, a source of food, fibre, materials for clothing, timber, and recreational opportunities. Biodiversity is also vital for the maintenance and enhancement of economic sectors (such as agriculture and tourism) during periods of rapid environmental change (Environment Canada, 2009).

This chapter summarizes the impacts of climate change on Canada's biodiversity and describes some of the tools available to maintain and enhance ecological resilience. Although most work to date has focused on the effects of climate change on individual species, some research on interspecific relationships is underway, which will enhance understanding of the impacts of climate change on ecosystem-level processes and ecosystem services (Walther, 2010).

Technical terms used in this chapter are defined in Box 1.

BOX 1

TECHNICAL TERMS USED IN THIS CHAPTER

Aerial insectivores: Bird species that eat insects while flying. Examples include the Chimney Swift and Barn Swallow.

Biogeochemical cycles: Pathways by which chemical elements or molecules move through both living and non-living compartments of an ecosystem.

Biodiversity: The variety of species and ecosystems and the ecological processes of which they are a part.

Biogeographical zones: Divisions of a land surface based on life form, species distribution, or the adaptation of plants and animals to climatic, soil, and other conditions.

Climate envelope: Model that predicts the distribution of a species in geographic space on the basis of a mathematical representation of its known distribution in a space represented by climate data (such as temperature, and precipitation).

Ecological integrity: A condition (of a protected area or other ecosystem) that is characteristic of its natural region and likely to persist, including non-living components and the composition and abundance of native species and biological communities, rates of change and supporting processes. Ecological integrity is a measure of ecological resilience (*see below*).

Ecological niche: The set of environmental conditions and resources that define the requirements of a species to complete its life cycle.

Ecological resilience: The capacity of a system to absorb disturbance and reorganize so as to still retain essentially the same function, structure and feedbacks, and therefore the same identity.

Ecological (or ecosystem) services: The multitude of resources and processes that are supplied by ecosystems and benefit human societies. These include products like clean drinking water and processes such as the decomposition of wastes.

Ecosystem: A community of living organisms (plants, animals and microbes) in conjunction with the nonliving components of their environment (air, water, soil), interacting as a system.

Hybridization: Interbreeding between two animals or plants of different species, subspecies or populations.

Hybrid vigour: Quality in hybrids of being stronger than either parent variety or species, a phenomenon most common with plant hybrids. Also called heterosis or heterozygote advantage.

Hypoxia: Oxygen depletion, a phenomenon that occurs in aquatic environments as dissolved oxygen becomes reduced in concentration to a point where it becomes detrimental to aquatic organisms living in the system.

Phenology: The description (or study) of periodic plant and animal life cycle events and how these are influenced by seasonal and yearly variations in climate, as well as habitat factors (such as elevation).

Zoonotic disease: An infectious disease that is transmitted between species (sometimes by a vector) from animals other than humans to humans or from humans to other animals.

Zooplankton: Heterotrophic (sometimes detritivorous) organisms drifting in oceans, seas, and bodies of fresh water.

2. OVERVIEW OF PREVIOUS ASSESSMENT FINDINGS

The ecological, social and economic effects of climate change were discussed in every regional chapter of '*From Impacts to Adaptation: Canada in a Changing Climate*' (Lemmen et al., 2008). Key conclusions related to biodiversity arising from that assessment include, but are not limited to:

- In Canada, northern ecosystems (e.g. taiga, tundra, and polar deserts) are, and will continue to be, particularly vulnerable to climate change. Impacts on Arctic species will involve habitat loss, competition from northward migrating species, and the arrival of new diseases and parasites from the south (Furgal and Prowse, 2008).
- Climate change impacts on species distribution, abundance, physiology and life cycle timing will alter interspecific relationships and habitats. Earlier onset of spring is changing the timing of growth and reproduction of many plant species that provide food and habitat for a variety of species. For example, the blossoming date of Trembling Aspen (*Populus tremuloides*) in Alberta has advanced by 26 days in the past 100 years (Beaubien and Freeland, 2000). Such timing shifts can cause decoupling of species that have co-evolved (Beaubien and Freeland, 2000 *in* Sauchyn and Kulshreshtha, 2008).
- Coastal and estuary ecosystems are at risk from increased erosion and "coastal squeeze" which could eliminate habitat for some species, such as the Piping Plover (Vasseur and Catto, 2008; Walker and Sydneysmith, 2008). Projected warming of the Gulf Stream and cooling of the Labrador Current could alter habitat and affect species in Atlantic Canada (Vasseur and Catto, 2008).
- Increased moisture stress in prairie ecosystems will likely decrease productivity in natural grasslands, although longer growing seasons and reduced competition from shrubs and trees (because of drier conditions) may partly offset the effects of reduced moisture (Sauchyn and Kulshreshtha, 2008).
- Habitats located in alpine ecosystems, cold steppes, and the Acadian forest could contract or become increasingly fragmented as the climate warms (Vasseur and Catto, 2008). Conversely, some species, including those of the Carolinian zone, may successfully occupy new niche space in more northern habitats as warming continues.
- Although forest productivity could increase with higher atmospheric CO_2 concentrations and longer growing seasons, increases in the frequency and intensity of fires, insect outbreaks, droughts, and icing events could offset potential gains. Eastward expansion of the mountain pine beetle epidemic into the boreal forest is also a concern (Bourque and Simonet, 2008).
- In Hudson Bay, observed changes in the distribution and abundance of seal and polar bear populations, as well as in a number of fish species, correlate with a decrease in the length of the sea ice season and higher water temperatures (Furgal and Prowse, 2008).
- Climate change impacts on water quantity and quality is a concern for lakes and rivers across Canada. Higher temperatures are affecting the thermal habitat of many fish species, increasing potential habitat for invasive species and creating favourable conditions for unwanted algal blooms (Chiotti and Lavender, 2008).
- In northern and alpine regions, the rapid melting of glaciers will change flow regimes with effects on downstream aquatic ecosystems and water supplies for many towns and cities (Sauchyn and Kulshreshtha, 2008).

Many examples of adaptation measures and approaches were also discussed in Lemmen et al. (2008). Measures to help protect biodiversity included: increased connectivity and reduced fragmentation between ecosystems; extension of protected area networks to conserve areas representative of each natural region; and implementation of inventory, monitoring and research programs to inform adaptive decision-making.

3. CLIMATE AND BIODIVERSITY

Climate is a key driver of ecosystem composition, structure, and function. It also interacts with other factors that influence biodiversity, such as pollution and land use change. As a result, many ecological studies include modeled or observed evaluations of relationships between biodiversity and climate change.

Current evidence indicates that climatically suitable ranges (or climate envelopes) for many species will likely shift northwards in response to warming temperatures (e.g. McKenney et al., 2007; Coristine and Kerr, 2011), with major implications for people who rely on the current configuration of ecosystem types. For example, ecological niche models for 765 species suggest that climate change may increase biodiversity in southern Quebec during this century as species move northward (Berteaux et al., 2010; Chambers et al. 2013). Similarly, many bird species that currently breed in the northern portion of eastern United States are likely to move northward into Canada, increasing bird species richness in Eastern Canada (Desgranges and Morneau, 2010).

Although migratory fronts may expand northward, the rear (southern) edge of species range is likely to contract in response to shifting climate (Hampe and Petit, 2005). Rear edges are often more genetically diverse than the rest of the population as a result of previous changes in species distributions (Jaramillo-Correa et al., 2009), such that the northward shift of southern populations may not only affect regional diversity and ecosystem function, but also the overall genetic diversity of the species involved (Hampe and Jump, 2011).

Biodiversity may also be affected when species shifts are limited by physical conditions (e.g. barriers to movement) and/or biological processes (e.g. reduced access to food at critical times in the life cycle such as breeding and rearing periods). Resulting changes in species composition can have varying consequences, such as disruptions in predator-prey and host-parasite relationships. Therefore, although we know that Canada's biodiversity will change in response to new climate conditions, uncertainties remain regarding how such changes will affect ecosystem composition, structure and function (Varrin et al., 2007). In all likelihood, response will be ecosystem/habitat-specific, resulting in a patchwork of increasing and decreasing species richness and productivity, changing with time across the country.

This section of the chapter addresses current understanding of the impacts of climate change on:

1. life cycle timing (phenology);
2. observed and projected species distribution;
3. fish and wildlife health; and
4. disturbance regimes and other human-induced stresses such as habitat fragmentation.

It also includes brief discussion of uncertainties, knowledge gaps and broader social and economic implications.

3.1 CHANGES IN THE TIMING OF LIFE-HISTORY EVENTS (PHENOLOGY)

Increases in winter and spring temperature at mid and high latitudes have led to earlier spring phenologies for some taxa (Schwartz et al., 2006a; Coristine and Kerr, 2011). Canadian examples of ongoing phenological shifts include earlier onset of breeding by amphibians (e.g. Walpole and Bowman, 2011; Walpole et al., 2012), earlier occupation of breeding habitat and emergence of hatchlings by bird species across North America (Waite and Strickland, 2006; Friends of Algonquin Park, 2012; Hurlbert and Liang, 2012; *see also* Case Study 1), and the earlier onset of the growing period for plant species (Schwartz et al., 2006a).

Insect physiology is strongly correlated with temperature. For example, in some insects, metabolism can almost double with an increase of 10°C, making them very responsive to temperature (Gillooly et al., 2001; Clarke and Fraser, 2004). Metabolic responses to increasing temperature influence, and likely amplify, population-level dynamics, including fecundity, survival, generation time and dispersal (Bale et al., 2002). Given that the timing of the spring life-cycle stages of many insect and plant species has already advanced in response to warmer temperatures (Harrington et al., 2001; Logan et al., 2003), a potential consequence for migratory birds is a phenological mismatch. This mismatch is characterized by a lack of synchrony between seasonal peaks in plant or insect biomass and hatchling growth and development (e.g. Rodenhouse et al., 2009; Knudsen et al., 2011; *see* Case study 1).

Changes in ocean climate (*see* Chapter 2 – An Overview of Canada's Changing Climate), particularly sea surface temperature, have a strong influence on the timing of life-history events for marine organisms. For example, in the Strait of Georgia, the period of peak abundance of the dominant species of zooplankton has progressed from occurring in late May fifty years ago to occurring in mid-March in 2004 (DFO, 2010). This change may also be a driver for the progressively earlier hatch dates for several seabirds in the area (Gaston et al., 2009) and is symptomatic of the strong links between recruitment success at higher trophic levels and the timing of temperature-dependent primary production in marine ecosystems (Bertram et al., 2009; Koeller et al., 2009). Species that have different habitat requirements for different life-history stages may be especially vulnerable to temperature changes. For example, juvenile salmon on both Atlantic (Friedland et al., 2003) and Pacific (Crozier et al., 2008) coasts are under increasing stress as a result of warming temperatures that may lead to a mismatch between the timing of smolting (a combination of behavioural, morphological, and physiological responses that stimulate migration and prepares the fish for life in the ocean) and biogeochemical conditions in the marine environment.

CASE STUDY 1

CLIMATE WARMING, PHENOLOGY MISMATCH AND POPULATION DECLINE IN MIGRATORY BIRDS

Birds are well studied and closely monitored, with research supported by long-term data sets assembled from many sources (e.g. Christmas Bird Counts, the Breeding Bird Survey). Analyses reveal sharp declines in some migratory bird populations, especially aerial insectivores, shorebirds and grasslands birds (Stuchbury, 2007; Nebel et al. 2010; Sauer et al., 2011; North American Bird Conservation Initiative Canada, 2012). In the last 40 years, 20 common North American bird species suffered declines in population levels of more than 50% (Federal, Provincial and Territorial Governments of Canada, 2010). Most declines are continuing, with thousands of birds lost every year. The rate and magnitude of decline has been high enough to warrant 'Threatened' status under the Species at Risk Act for species such as the Common Nighthawk (*Chordeiles minor*), Chimney Swift (*Chaetura pelagica*), Canada Warbler (*Cardellina canadensis*), Eastern Meadowlark (*Sturnella magna*) (COSEWIC, 2007a & b, 2008, 2011). Similar declines of migratory bird populations in recent decades have been documented in Europe (Møller et al., 2008).

Climate change is likely an important factor in current population declines (Knudsen et al., 2011; North American Bird Conservation Initiative Canada, 2012), contributing to habitat deterioration on wintering grounds of migratory birds due to drought and other climate impacts, and an increasing phenological mismatch – a decoupling of the timing of migration and high food abundance. While there is evidence that phenological mismatches can have significant effects on populations and species (Post et al., 2009; Knudsen et al., 2011; Miller-Rushing et al., 2012), few studies have explicitly addressed the effects of climate change on phenological mismatches for migratory birds.

An index of phenological mismatch, calculated as the difference between temperature trends on wintering grounds compared to breeding grounds, is a good predictor of population declines in North American bird populations (Jones and Cresswell, 2010). In Europe, bird species that did not advance their spring migration declined during 1990 to 2000, whereas species that did advance the timing of migration had stable or increasing populations (Møller et al., 2008). Also, despite earlier arrival dates, birds now arrive on higher degree-days than in the past so that heat-dependent ecological processes, such as insect emergence, have advanced relatively further, creating a 'thermal delay' (Saino et al., 2010, Figure 1). Bird species with greater 'thermal delay' have experienced steeper population declines.

Some European species (including the Blue Tit (*Parus caeruleus*), Great Tit (*Parus major*), Pied Flycatcher (*Ficedula hypoleuca*), Sparrowhawk (*Accipiter brevipes*) demonstrated increased mismatch due to breeding dates advancing either less (Both et al., 2009) or more (Pearce-Higgins et al., 2005) than the advancement of the main food peak available to nestlings. However, the phenological mismatch may not occur in environments with relatively abundant food throughout the breeding season. For instance, in North America, the onset of egg laying by the aerial foraging Tree Swallow is strongly correlated to flying insect biomass during the laying period and not to the timing of the seasonal peak in food supply, which occurs later in the season in most sites and years (Dunn et al., 2011).

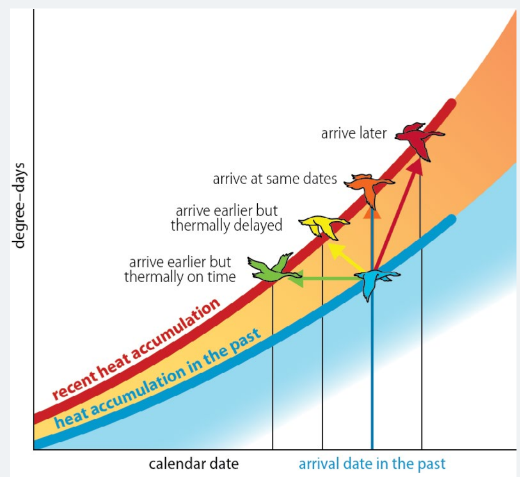

FIGURE 1: Climatic and phenological changes can bring about ecological mismatch of migratory birds. Curves represent the progress of spring in two years, as the increase of degree-days (heat accumulation) over time. The curve for the recent year (red line) lies above that for the past (blue line) because of winter and spring warming, which means that degree-days increase more rapidly. Migratory birds show no change, advancement or delay in arrival date. Species that now arrive at the same or later date face higher degree-days and relatively advanced ecological processes such as insect emergence, and are thus 'thermally delayed'. Even species that have advanced their arrival may experience a thermal delay, if advancement does not fully compensate for increasing temperatures. Only a large advancement in arrival can fully compensate for climate change *(modified from Saino et al., 2010)*.

3.2 CHANGES IN SPECIES DISTRIBUTION RANGES

There is strong observational evidence of historic species range shifts. Over the past 40 years, about 180 of 305 bird species wintering in North America expanded their range northward at an average rate of 1.4 km per year. Similarly, the breeding ranges of birds in southern North America have shifted by an average of 2.4 km per year (Federal, Provincial and Territorial Governments of Canada, 2010). Within the northeastern forests of North America, 27 of 38 species for which historical ranges are documented have expanded their ranges predominantly northward (Rodenhouse et al., 2009). Published accounts of range shifts in Canada are available for a number of species (Hitch and Leberg, 2007; Blancher et al., 2008), with detailed analyses for the Hooded Warbler (*Setophaga citrina*) (Melles et al., 2011), the Southern Flying Squirrel (*Glaucomys volans*) (Garroway et al., 2010; 2011), butterflies (Petersen et al., 2004; Kharouba et al., 2009), and a number of tree species (Gamache and Payette, 2005; Asselin and Payette, 2006; Crête and Marzell, 2006; Boisvert-Marsh, 2012).

Future range shifts are commonly projected using species distribution models or climate envelope models. These models use the statistical correlations between known occurrences of species or ecosystem types and the climate variables associated with those occurrences (e.g. Thuiller et al., 2005; Hamann and Wang, 2006) to predict distributions under projected future (or past) climate conditions (Thuiller et al., 2005; Berteaux et al., 2010, 2011; Pellatt et al., 2012). Currently, most models assume that climate is the primary determinant of habitat suitability. This assumption may be valid on broad geographic scales, but is likely less valid at smaller spatial scales (*see* section 3.5). Models project range expansion when the spatial extent of suitable climate increases, but when species have geographically limited ranges, warming may result in range contraction. Many arctic and alpine species are expected to undergo range contraction in response to warming (Alsos et al., 2012) as opportunities for range expansion up-slope or northward may not exist.

Changes in species distributions under future climate scenarios also depend on whether the capacity of the species of interest to disperse is taken into account. For example, under a scenario in which there was no limit to the ability of seeds to colonize new habitat, projections of future distributions for 130 North American tree species were for a 12% range contraction and an average 700 km northward shift of the centre of distribution by the end of the century. However, when it was assumed that trees could not migrate through seed dispersal, a 58% range contraction and a 330 km northward shift were projected (McKenney et al., 2007). The most likely outcome may lie between the two extreme dispersal scenarios, particularly in Northern Canada where the lack of fertile soil is likely to limit the northward migration of many tree species.

Major changes in forest species composition are expected under most climate projections for eastern North America, including a reduction in the area suitable for many northern hardwood species (Iverson et al., 2008). Northern hardwood species are likely to be replaced by species characteristic of an oak–hickory forest type and various pine species, but the rate of change is uncertain (Iverson et al., 2008). Local or regional soil properties are likely to slow down potential tree migration (Lafleur et al., 2010). Some research suggests that we may expect continually changing forest tree communities in the future, with tree species responding in different ways to climate variables and soil properties (Drobyshev et al., 2013).

Scenarios based on bioclimate envelope models for western Canada indicate that tree species with their northern range limit in British Columbia could expand their range at a pace of at least 100 km per decade (Hamann and Wang, 2006). While common hardwoods appear to be generally unaffected by changes in average temperature and precipitation, some economically important conifer species in British Columbia, such as Lodgepole Pine, could lose a large portion of their range (Figure 2a). It is important to note that it is unlikely that all of the properties that define a climate envelope of a species will shift together. The potential redistribution of biogeographical zones is considerable, with currently important sub-boreal and montane climate regions predicted to reduce rapidly (Figure 2b).

Animals that breed in high-elevation forests are highly vulnerable to climate change because there is little opportunity to shift to new habitat at higher locations. For example, models project that the Bicknell's Thrush will lose access to a significant area of breeding habitat with a 1°C increase in mean annual temperature (Lambert et al., 2005; Rodenhouse et al., 2008).

Range expansion and contraction can have genetic consequences for species populations. While the dispersal of individuals can increase local genetic diversity and spread beneficial genotypes (Hewitt and Nichols, 2005), genetic diversity can decrease at the outer boundaries of a species' distribution when few populations colonize new habitat. Genetic diversity also decreases when a population is extirpated at the contracting edge of the species range (Hewitt and Nichols, 2005; Hampe and Jump, 2011). Models for 27 common arctic plant species project decreases in genetic diversity in response to climate-related reductions in

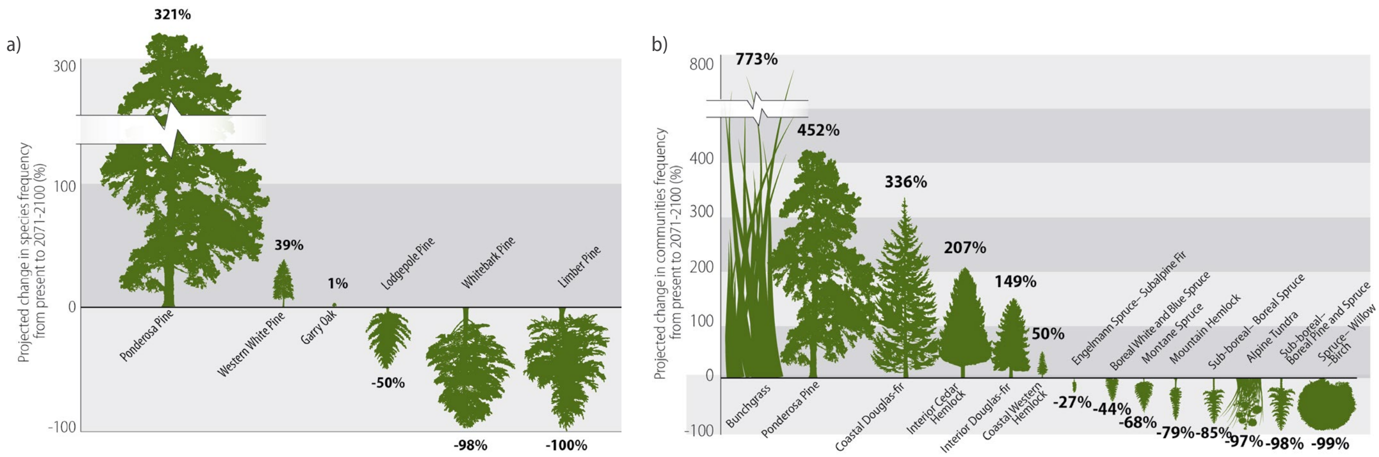

FIGURE 2: Projected species and community changes from present to 2071-2100 in British Columbia in response to climate change *(modified from Hamann and Wang, 2006)*. **a)** Change in the frequency of pine and oak species, with Ponderosa Pine (*Pinus ponderosa*) becoming much more common, while Whitebark Pine (*Pinus albicaulus*) and Limber Pine (*Pinus flexilis*) could disappear. **b)** Change in the distributions of biogeoclimatic ecological zones, with Bunchgrass communities and Ponderosa Pine forests becoming much more common than today, while alpine tundra, for instance, may disappear.

range size (Alsos et al., 2012). Analysis of *in situ* observations are beginning to appear in the published literature as well. For example, reductions in genetic diversity have been documented for the alpine chipmunk (*Tamia alpinus*) in response to climate-related upslope range contraction (Rubidge et al., 2012).

Climate change-related range expansion may increase the likelihood of interbreeding and hybridization when two previously distinct populations or species come into contact (Hoffmann and Sgrò, 2011). For example, the Southern Flying Squirrel (*Glaucomys volans*) expanded its northern range by approximately 200 km (Bowman et al., 2005; Garroway et al., 2010, 2011) during a series of warm winters in Ontario between 1994 and 2003. This range expansion resulted in increased co-occurrence and hybridization between the Southern and the Northern Flying Squirrel (*Glaucomys sabrinus*) (Garroway et al., 2010). Hybridization can result in: i) sterile offspring; ii) viable offspring with increased fitness; iii) viable offspring with reduced fitness, or iv) no change in fitness. In some cases, hybridization may facilitate capacity of some species to respond to change by introducing genetic variation (Hoffmann and Sgrò, 2011).

Hybridization can lead to extinction if a species is rare and hybridizes with a more abundant species. For example, the plant Lesser Pyrola (*Pyrola minor*) is locally rare at a number of locations along its southern range margin, and genetic studies of this species have revealed that hybridization with Arctic Pyrola (*Pyrola grandiflora*) is resulting in the increased presence of hybrids at the expense of *P. minor* (but not *P. grandiflora*) in Greenland and Northern Canada. This process may lead to the extinction of *P. minor* through genetic assimilation (Beatty et al., 2010). Similarly, a common North American songbird, the Black-capped Chickadee (*Poecile atricapillus*) hybridizes with the Carolina Chickadee (*Poecile carolinensis*) (Curry, 2005), with mixed parents having reduced hatching success and their offspring having reduced reproductive success (Varrin et al., 2007). A narrow area of reduced reproductive success was detected within the current hybrid zone between these species (Bronson et al., 2005). Given that the Carolina Chickadee is steadily expanding its range northward (Hitch and Leberg, 2007), the hybrid zone could expand from Ohio into Ontario within a few years. Eventually, hybrid chickadees could replace the Black-capped Chickadee in southern Ontario.

Hybridization may increase an organism's capacity to cope with climate change (Hoffmann and Sgrò, 2011). For example, Fishers (*Martes pennanti*) in Ontario demonstrate hybrid vigour between recolonizing populations (Carr et al., 2007), and the emergence dates of budworm species (*Choristoneura* spp.) are more responsive to spring temperature change when they hybridize with a closely related species of the same genus (Volney and Fleming, 2000).

Changes in the distribution of fish and other aquatic organisms are also occurring and/or projected to occur. Although the distribution of cold-water fish (e.g. Lake Trout [*Salvelinus namaycush*]) is restricted by warm summer temperatures (Rahel, 2002), physical limnology and lake geography also affect the habitability of northern waters (Minns et al., 2009). Projections of the effects of climate change on Lake Trout habitat in Ontario suggest that by 2100, Lake Trout habitat will be reduced by 30%, with declines of up to 60% in some southern watersheds. These declines would be only partly offset by habitat increases in watersheds of northwestern Ontario (Minns et al., 2009).

The northern limits of warm-water fish species is often determined by cold summer temperatures, which limit growth (Shuter and Post, 1990; Rahel, 2002). For example, waters with temperatures over 27°C and/or below 15°C do not provide adequate thermal habitat for Smallmouth Bass (*Micropterus dolomieu*), such that only lakes in more southerly locations of Canada currently provide suitable thermal habitat. However, climate projections suggest that significantly more lakes in more northerly locations will provide suitable habitat for warm-water species by the end of the century (Chu et al., 2005). Specifically, the northern range of Smallmouth Bass habitat may expand into northwestern Ontario, northeastern Manitoba, and south-central Saskatchewan, disrupting existing food webs. Jackson and Mandrak (2002) suggest that such range expansions could lead to the localized extinction of 25 000 populations of Northern Redbelly Dace (*Phoxinus eos*), Finescale Dace (*Phoxinus neogaeus*), Fathead Minnow (*Pimephales promelas*) and Pearl Dace (*Margariscus margarita*). Such extirpation would negatively affect food resources for predators such as Lake Trout (Vander Zanden et al., 1999). Barriers to migration for aquatic species, including constructed barriers and watershed divides, along with improved public understanding of risks associated with transporting species between water bodies, could impede those projected range expansions.

Marine species may respond to warmer temperatures by altering their depth and latitudinal ranges (*see* Chapter 4 – Food Production). Estimates suggest latitudinal ranges may shift northward by 30 to130 km per decade, while depth ranges may shift 3.5 m deeper per decade (Cheung et al., 2009, 2010). In Canada, species gains and losses are both predicted for marine habitats, with the greatest losses occurring at lower latitudes. However, the overall number of species in Canadian waters is likely to increase (Cheung et al., 2011). Rapid biogeographical changes are already evident, with warm-water species moving northwards more than 10 degrees of latitude over the last 30 years in the North Atlantic, along with concomitant declines in the diversity of cold-water species (Helmuth et al., 2006). A warming Arctic Ocean is expected to result in an expansion of Pacific species into the Arctic, and from there into the Atlantic Ocean (Vermeij and Roopnarine, 2008; *see also* Reid et al., 2008). Other range expansions may not be as dramatic, but could have significant impacts on coastal communities. For example, Harley (2011) documents how the Sea Star, *Pisaster ochraceus*, is favoured by warming conditions and suggests that this will reduce the areal extent of the mussel beds on which it preys, with related impacts on other species associated with mussel beds.

3.3 EFFECTS ON FISH AND WILDLIFE HEALTH

A number of studies document the effects of climate change on the health of fish and wildlife species in Canada. Examples of impacts on select iconic species are highlighted here.

Few species are more linked by the public to climate change than polar bears. Polar bears depend on sea ice for hunting and mating – they gain weight in April, May, and June, just prior to ice break-up when newly weaned ringed seals are abundant (Stirling et al., 1999; Derocher et al., 2004; Rosing-Asvid, 2006). Polar bears, particularly southern populations, will be affected by changes in the timing of ice break-up and freeze-up and the formation of the ice pack. As ice platforms disintegrate earlier and form later, polar bears will have less time to feed on seals, which will result in poorer condition of reproductive females and lower reproductive success (Stirling et al., 1999; Obbard et al., 2006; Peacock et al., 2011). These impacts are only part of the complex suite of ecological changes that have already been documented in the Arctic (*see* Case Study 2).

CASE STUDY 2

CLIMATE-DRIVEN ECOLOGICAL CHANGE IN THE ARCTIC

A summary of recent research on environmental change in arctic national parks (McLennan et al., 2012) shows that these relatively pristine ecosystems are undergoing significant and accelerating change in the cryosphere (glaciers, permafrost, snow cover, lake, river and sea ice) and vegetation, while wildlife populations are just beginning to respond to the effects of warming that is more than double that of southern latitudes (*see* Chapter 2 – An Overview of Canada's Changing Climate). National parks and other protected areas can act as 'benchmarks of change' that help us understand the nature, magnitude and rate of change occurring in natural systems as a result of climate change (Lemieux et al., 2011; CPC, 2013).

Derkson et al. (2012) provide a comprehensive review of recent changes in the cryosphere of the Canadian Arctic. Glaciers in arctic national parks are receding and losing mass (Dowdeswell et al., 2007; Barrand and Sharp, 2010; Gardner et al., 2011; Sharp et al., 2011), while permafrost is warming and the depth of summer soil thaw is increasing (Burn and Kokelj, 2009; Smith et al., 2010). Increasing summer thaw depth is an important factor contributing to increased incidence of landslides (Broll et al., 2003). Permafrost degradation in ice-rich ground allows wetlands to drain and changes the water regime of tundra ecosystems (Fortier et al., 2007; Godin and Fortier, 2012).

Vegetation changes observed in arctic national parks include differential growth of individuals of species already present at a given location, as well as range expansions of Low Arctic trees and shrubs, resulting in significant changes in tundra community structure. Tundra vegetation is growing more quickly, a phenomenon correlated with sea ice reduction (Lawrence et al., 2008; Bhatt et al. 2010; Stroeve et al., 2011), and is resulting in an overall 'greening' across the north (Jia et al., 2009). In four arctic national parks, shrub coverage is expanding and herbaceous species are increasingly occupying previously bare ground (Fraser et al., 2012). In Ivvavik National Park, Yukon, seasonal peak leaf biomass more than doubled between 1985 and 2010 (Figure 3), and the growing season increased on average by over 40 days in the same period (Chen et al., 2012). Results suggest that warming is increasing plant growth in arctic national park ecosystems, but response varies among species and sites (Hill and Henry, 2011).

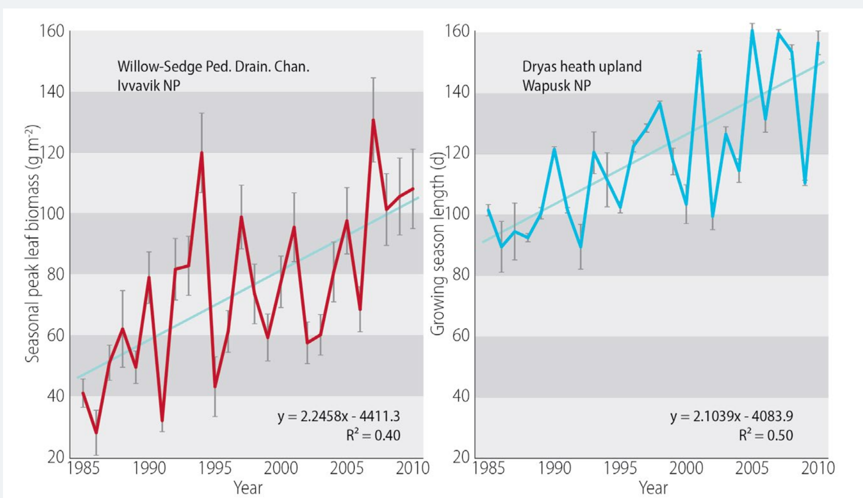

FIGURE 3: Seasonal peak leaf biomass (g m^{-2}) more than doubled between 1985 and 2010 in the Willow-sedge pediment drainage channel ecotype in Ivvavik National Park, Yukon, and growing season length almost doubled in Dryas heath upland in Wapusk National Park, Manitoba *(Chen et al., 2012)*.

Vegetation changes can have impacts ranging from local effects on habitats and soil processes (Sturm et al., 2005a; Post et al., 2009), to watershed scale effects on hydrology (McFadden et al., 2001), to global effects on climate through carbon cycle feedbacks (Sturm et al., 2005b; Ping et al., 2008; Bonfils et al., 2012).

Animal populations are also changing, with potential consequences for arctic food webs. For example, research on food webs in Sirmilik National Park, Nunavut (Figure 4) demonstrates that lemmings play a keystone role supporting arctic biodiversity because of their widespread abundance and a consequent role as prey for many arctic raptors and mammalian predators (Gauthier et al., 2004, 2011;

Case Study 2 continued on next page

Gauthier and Berteaux, 2011; Therrien, 2012). While there is strong evidence in northern Europe and Greenland that climate change is impacting lemming numbers and the predators that rely on them (Kausrud et al., 2008; Gilg et al., 2009), there is little evidence for this so far in Canada (Gauthier et al., 2011; Krebs et al., 2011). Difficulties inherent in projecting changes in snow depth and phenology, and the natural local-scale variability in snowfall, create a key uncertainty in predicting the future of arctic small mammals and the species they support.

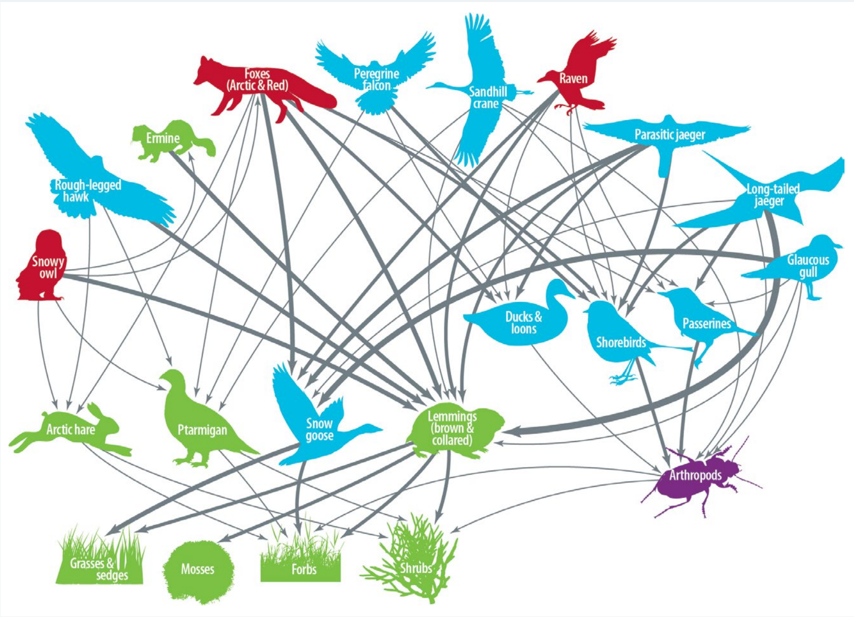

FIGURE 4: Bylot Island (Sirmilik National Park) food web showing four species categories: migrant (red), partial migrant (blue), resident, (green), and resident frozen (purple) in the soil during winter. Line thickness indicates the relative strength of interactions between species *(modified from Gauthier et al., 2011)*.

Arctic national parks also support key breeding ranges of several barren ground caribou herds. Caribou numbers are presently decreasing for most herds across the circumpolar arctic (CARMA, 2012). Population fluctuations in barren ground caribou are interpreted to be a natural cycle caused by the interaction of long-term climate cycling with forage quality and calf survival (Payette et al., 2004; Sharma et al., 2009). However, the ability to recover from low numbers may now be affected by climate-related stresses such as changes in vegetation phenology (White, 2008; Sharma et al., 2009), increases in icing events and insect harassment (Sharma et al., 2009, CARMA, 2012), and increases in forest fire frequency on wintering grounds (Joly et al., 2012). These climate-related factors are exacerbated by a host of increasing anthropogenic stressors (Sharma et al., 2009; CARMA, 2012, Joly et al., 2012; *see* Section 3.4). Negative factors need to be balanced against the potentially positive effects of increased biomass of caribou forage, and overall warmer temperatures (Griffith et al., 2002). The interaction of all of these factors creates uncertainty around the future of the large barren ground caribou herds that are iconic features of arctic national parks, key drivers of arctic biodiversity and ecosystem processes, and central to community culture and land-based lifestyles. Similar uncertainty exists around Peary Caribou (*Rangifer tarandus pearyi*) – a species whose populations have been negatively impacted by extreme climate events (Miller and Gunn, 2003), but which may recover if increases in forage productivity outweigh the negative effects of climate change (Tews et al., 2007).

Some southern populations of polar bears, in particular the western Hudson Bay (Wapusk National Park, Manitoba) and southern Beaufort Sea (Ivvavik National Park, Yukon) populations, have been projected to disappear over the next 30-40 years (Stirling and Derocher, 2012). It can be expected that reduced sea ice will force populations north as summer sea ice reductions continue. Although significant changes in arctic fauna are inevitable, these changes are only just beginning to occur. In the next few decades it can be expected that more mobile subarctic and boreal species of songbirds, raptors, small mammals, ungulates, and predators will move north, creating complex interactions with arctic species already present (ACIA, 2005; Berteaux et al., 2006; Gilg et al., 2012). The immigration of southern species through range shift or expansion associated with a warming climate will put pressure on obligate arctic species such as polar bears, muskox, lemmings, arctic fox and Peary caribou (Berteaux et al., 2006; Berteaux and Stenseth, 2006; Gilg et al., 2012). Areas of the Arctic with mountainous terrain, which includes several national parks, have the potential to provide climate refugia at elevation or on north-facing slopes. Understanding the potential role of refugia in harbouring species with contracting ranges, and the potential consequences for species restricted to such refugia (Barnosky, 2008; Keppel et al., 2012) will be important for predicting the fate of many obligate arctic species.

Overall, evidence from monitoring and research in national parks indicates a high level of uncertainty regarding how terrestrial ecosystems in Canada's north will respond to climate change. This uncertainty relates in part to difficulties in predicting how climates will change at spatial and temporal scales relevant to management action (ACIA, 2005; McLennan, 2011), and to the inherent complexity of natural ecosystems (Berteaux et al., 2006; Gilg et al., 2012). Managers of arctic national parks and communities who are dependent on arctic ecosystems for food supplies or livelihoods are presented with a situation where radical ecological change is inevitable.

In aquatic ecosystems, water temperature is a principal determinant of fish survival and reproduction. Mean summer water temperatures in the Fraser River of British Columbia have increased by about 1.5°C since the 1950s (Martins et al., 2011). Record high river temperatures during recent spawning migrations of Fraser River Sockeye Salmon (*Oncorhynchus nerka*) have been associated with high mortality events, raising concerns about the long-term viability of natal stocks under emerging climate regimes. Analysis of four Fraser River Sockeye Salmon stocks show varying responses, with some, but not all, affected by warm temperatures encountered in the lower river (Martins et al., 2011). A decrease of 9 to16% in survival of all of the stocks is predicted by the end of the century if the Fraser River continues to warm as projected. The study emphasized the need to integrate consideration of stock-specific responses to temperature changes into fisheries management and conservation strategies.

In marine ecosystems, changing climate is associated with a range of physiological stresses that affect species health. For example, Crawford and Irvine (2009) attribute documented hypoxia (declines in dissolved oxygen) at all depths below the mixing layer along the entire BC coast to warming waters off the coast of Asia that increase stratification and reduce ventilation (Whitney et al., 2007, *see also* Chapter 2 – An Overview of Canada's Changing Climate). Increasing greenhouse gas emissions are also increasing ocean acidification, with far-reaching implications for the long-term distribution and abundance of marine species (Feely et al., 2008, Friedlingstein et al., 2010). For example, the availability of calcium carbonate, the building block for construction of carbonate skeletal structures and hard shells by marine organisms, has declined as a result of ocean acidification. Cold northern waters absorb carbon dioxide more efficiently than southern waters, and warmer summers promote faster melting of the more acidic sea ice (Yamamoto-Kawai et al., 2009). Acidification has also been documented off the west coast of Canada (Feely et al., 2008; Cummins and Haigh, 2010) and in the Gulf of St Lawrence, where dramatic falls in pH are also linked to increased hypoxia (Dufour et al., 2010).

3.4 INTERACTIONS WITH DISTURBANCE REGIMES AND HUMAN-INDUCED STRESSES

Significant synergies are likely amongst the many ecological and socio-economic pathways affected by climate change. For example, Ainsworth et al. (2011) concluded that for marine environments of the northeastern Pacific Ocean, the combination of deoxygenation, acidification, primary production, zooplankton community structure and species range shifts were more severe than the sum of the individual impacts. Furthermore, climate change impacts interact with other human-induced and natural stresses, including habitat loss and fragmentation, pollution, over-harvesting, forest fire, and invasive species, such that the cumulative effects could threaten many species (Venter et al., 2006).

The cumulative effects of climate change and habitat fragmentation will likely limit adaptation success in some species (Travis, 2003; Opdam and Wascher, 2004; Inkley et al., 2004; Bowman et al., 2005; Varrin et al., 2007). Successful colonization of forest plants is higher in more connected landscapes, and plant species dispersed by animals are better able to colonize new habitats than those dispersed by other means (Honnay et al., 2002). Fragmented landscapes may also be an impediment to range expansion by birds, particularly smaller local populations at the limits of physiological tolerance (e.g. Opdam and Wascher, 2004; Melles et al., 2011). Mammals such as the Southern Flying Squirrel, which is a forest-obligate that has spread north through the contiguous forested habitats of the Precambrian Shield in eastern Ontario, but not through the fragmented forests of the southwest (Bowman et al., 2005), may also be affected. Habitat fragmentation is also a significant issue in freshwater ecosystems (Allan et al., 2005), particularly with respect to dams, diversions, revetments, lotic reaches replaced by reservoirs, channel reconfiguration, or dewatering (Stanford, 1996).

Other human-related stressors, including water pollution, wetland drainage and lowering of ground water tables have significantly degraded freshwater ecosystems in much of North America over the past 60 years (Kundzewicz and Mata, 2007) and climate change will exacerbate many of these effects. Similarly, harvesting in marine ecosystems has led to changes in the composition and abundance of fish communities in Canada's waters (*see* Chapter 4 – Food Production). These changes, along with climate-driven changes in temperature, salinity, and acidity, have led to significant changes in Canadian marine biodiversity (Benoît and Swain, 2008; Templeman, 2010).

Fire and insect outbreaks are natural drivers of ecosystem change throughout most of Canada. Climate change, however, is projected to alter the frequency and magnitude of these disturbances (*see* Chapter 3 – Natural Resources). The average area burned annually in Canada is projected to increase by 75 to 120% by 2100 (Flannigan et al., 2009; Stocks and Ward, 2011). Effects will vary regionally – for example, the average area burned per decade across Alaska and western Canada is projected to double by 2041-2050, compared to 1991-2000, and could increase up to 5.5 times by the last decade of the 21st century (Balshi et al., 2009).

Invasive species are those beyond their natural range or natural zone of potential dispersion (Mortsch et al., 2003) that can potentially cause harm by outcompeting, preying upon, or parasitizing indigenous species (Varrin et al., 2007). Many of the invasive aquatic species currently inhabiting

Canada will respond positively to warmer water temperatures, with significant implications for ecosystem health and parts of the economy. For example, many invasive species from Europe (e.g. Ponto-Caspian species) and Asia originated in warmer waters, which can give them a competitive advantage over cool- and coldwater indigenous species under climate warming (Schindler, 2001). Asian Carp species are currently thriving in the Mississippi River system and have reached waters as far north as the Chicago Sanitary and Ship Canal. Although an electric barrier currently prevents carp from entering Lake Michigan, the potential for invasion is considered significant. In addition, attempts to transport live carp into Canada have occurred in the past and will potentially occur in the future (Dove-Thompson et al., 2011). In the ocean marine environment, the European Green Crab (*Carcinus maenas*) has moved north from California into the waters of British Columbia (Klassen and Locke, 2007), and two tunicates (*Botryllus schlosseri* and *B. violaceus*) that are now found on Vancouver Island have the potential to invade most of coastal BC under warming conditions (Epelbaum et al., 2009).

While invasive plant species are commonly thought to disproportionately benefit from climate change, compared to native species, this has not been directly tested (Dukes et al., 2009). Several studies suggest that successful invasive plant species tend to have broad environmental tolerances (Goodwin et al., 1999; Qian and Ricklefs, 2006) and other characteristics that allow them to maintain or increase their fitness relative to other species in a changing climate. Such characteristics include short generation times, long and frequent periods of propagule (e.g. seed) dispersal, and traits that facilitate long-range dispersal which equip many invasive species to outcompete indigenous species that are less well-adapted to new climates (Pitelka et al., 1997; Dukes and Mooney, 1999; Malcolm et al., 2002).

Warming can also significantly affect life cycle dynamics and the distribution of indigenous eruptive insects such as the Mountain Pine Beetle (*see also* Chapter 3 – Natural Resources). Warming simulation models suggest that increases in mean temperature of 1°C to 4°C will significantly increase the risk of outbreaks, starting at higher elevations (for increases of 1°C and 2°C) and then increasing at more northern latitudes (4°C) (Sambaraju et al., 2012). In the Colorado Front Ranges, the flight season of Mountain Pine Beetle now begins more than one month earlier than reported in the past, and lasts approximately twice as long (Mitton and Ferrenberg, 2012). The life cycle in some broods has also increased from one to two generations per year. Because the Mountain Pine Beetle does not go dormant, and growth and maturation is controlled by temperature, this species is responding to climate change through faster growth (Mitton and Ferrenberg, 2012). Furthermore, the Mountain Pine Beetle has higher reproductive success in areas where its host trees have not experienced frequent beetle epidemics (Cudmore et al., 2010). This increased fecundity may be a key reason for the rapid population explosion that resulted in unprecedented host tree mortality over large areas in western Canada (*see* Chapter 3 – Natural Resources). Forest management practices, such as maintenance of a mosaic of species and age classes at the landscape level can help reduce the impacts of such outbreaks (Cudmore et al., 2010).

3.5 UNCERTAINTIES AND KNOWLEDGE GAPS

Although the evidence for climate-driven biodiversity change in Canada is strong, predicting and measuring such change is laden with uncertainty. The expectation that species will shift their distributions as a result of climate change generally assumes that climate is the primary influence on habitat suitability. While climatic factors drive diversity patterns at large scales and define the range of most species, many biophysical factors and interactions such as competition, predation and symbiosis also influence population distribution and abundance at local scales (McLachlan et al., 2005; Anderson and Ferree, 2010).

Overall, the extent to which species can achieve rapid large-scale migrations is still poorly understood (Pearson, 2006). The use of climate envelope models to estimate potential effects on species distribution is limited because such models are not integrated with population models that help identify extinction risks (Brook et al., 2009). For example, climate envelope models that average climate over large areas do not incorporate localized microclimates within which low-density populations can persist (e.g. *see* McLachlan et al., 2005), and may therefore be too coarse to incorporate key mechanisms by which species can persist through rapid changes in climate (Pearson, 2006). The ability of climate envelope models to predict current ranges of North American trees and birds declines with range size, and climatic parameters become less important explanatory variables for small ranges (Schwartz et al., 2006b). As a result, the extinction risk of narrowly distributed species might not be well predicted, even though sparse and endemic species are important components of predicting the overall extinction risk brought by climate change. It is also important to note that even though a species may exist within suitable climate space, factors such as competition, food availability, disease, and predation may play a more important role in whether or not they persist in an area.

Some limitations have been addressed, in part, by integrating climate with population models. Such models provide insights on how the complex interactions between life history, disturbance regime and distribution influence the increased risk of extinction for a given species under climate change (Keith et al., 2008). For instance, the levels of dispersal required to offset movement of climate envelopes were found to be beyond the biologically plausible bounds of dispersal for some plant species. Despite these advances, continued research into the effects of climate change on species "niche" patterns and response of species groups or assemblages is important to understanding the reconfiguration process of Canada's ecosystems as temperatures continue to warm.

3.6 SOCIAL AND ECONOMIC IMPLICATIONS OF CLIMATE-DRIVEN BIODIVERSITY CHANGE

It is well recognized that biodiversity provides ecosystem services (and goods derived from ecosystems) that ensure human well-being (Federal, Provincial and Territorial Governments of Canada, 2010). In Canada, ecosystem services contribute significantly to a range of economic sectors, including agriculture and fisheries (*see* Chapter 4 – Food Production), forestry (*see* Chapter 3 – Natural Resources) and tourism (*see* Chapter 5 – Industry). For example, biodiversity serves an important function in pollination and forest productivity (Thompson et al., 2011). Such services may be directly provided by one or more species within a community (e.g. long-term carbon sequestration by *Sphagnum* in a peat bog; *see also* McLaughlin and Webster, 2013), by the presence of rare species in a community (providing resistance to invasive species; Lyons and Schwartz, 2001) or through interactions between multiple species in an ecosystem (Cardinale et al., 2011). Species richness and abundance are key determinants of ecosystem function (Hooper et al., 2012), including the provision of ecosystem services.

Biodiversity is also linked to human health and well-being (*see* Chapter 7 – Human Health). Declines in local and regional biodiversity have been tied to increasing rates of allergies in adolescents (Hanski et al., 2012), and may also increase rates of zoonotic disease, such as the West Nile Virus and Lyme disease in humans (Ostfield, 2009). Declines in biodiversity result in declines in potential host taxa with low suitability for the pathogens. These low-suitability hosts are replaced by generalists such as the American Crow (*Corvus brachyrhynchos*) and Blue Jay (*Cyanocitta cristata*) for the West Nile virus and the White-footed Mouse (*Peromyscus leucopus*) and Eastern Chipmunk (*Tamias striatus*) for Lyme disease (Ostfield, 2009). Both diseases are expected to expand their distributions under changing climate regimes (Hongoh et al., 2011).

The economic valuation of ecosystem services is a relatively new and complex discipline. Estimates have been prepared for a number of areas in Canada where organizations are exploring ways of integrating these values into decision-making processes. The value of provisioning (market) services in Canada's boreal forest, for example, has been estimated at $37.8 billion per year, while non-market ecosystem services, including pest control and nature-related activities represent another $93.2 billion per year in services (Anielski and Wilson, 2009). The proposed Rouge National Urban Park and its surrounding watersheds are estimated to provide $115.6 million annually ($2247 per hectare per year) in non-market economic benefits for residents in the Greater Toronto Area (Wilson, 2012). The ecosystem services that contribute most to the total study area's natural capital assets are pollination services ($28.2 million per year), stored carbon ($17.8 million per year), and wetland habitat ($17.1 million per year).

The linkages between biodiversity, ecosystem services and climate change highlight the importance of ecological resilience as a foundation for societal adaptation in many areas (e.g. SCBD, 2009; Staudinger et al., 2012; Hounsell, 2012; Munang et al., 2013). Decreased quantity, quality and access to ecosystem services, which is the result of many factors and aggravated by climate change (Mooney et al., 2009; Federal, Provincial and Territorial Governments of Canada, 2010; Hounsell, 2012) increases the vulnerability of resource-dependent communities (Figure 5; Vasseur, 2010; Klein et al., 2011). In Canada, Aboriginal communities that rely on traditional (wild) food supplies are especially vulnerable to changes in species and ecological processes (*see* Chapter 4 – Food Production). This is particularly true in the North, where climate-related impacts are expected to be greatest.

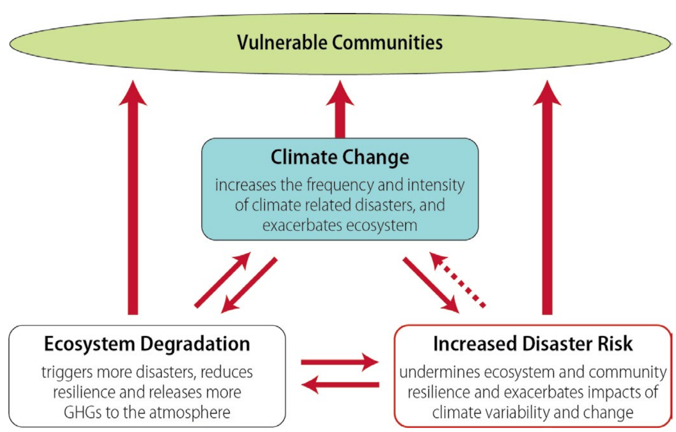

FIGURE 5: Illustration of the linkages between climate change impacts, ecosystem degradation, and increased risk of climate-related disasters *(modified from UNEP, 2009).*

3.7 SYNTHESIS

The impact of human-caused stressors, including climate change, on Canada's biodiversity is evident. Changes include the loss of old-growth forests, changes in river flows at critical times of the year, loss of wildlife habitat in agricultural landscapes, declines in some bird populations, increases in wildfire, significant shifts in marine, freshwater, and terrestrial food webs, and the increasing occurrence of some contaminants in wildlife (Federal, Provincial and Territorial Governments of Canada, 2010). Responses of species and ecosystems to climate change interact in complex ways such that an impact on one component can have cascading effects on others (Figure 6), leading in some cases to the transformation to new interspecific relationships, and to ecosystems with new characteristics (e.g. Gray, 2005). Climate change poses an additional stress on ecosystems and species that may already be reaching critical thresholds, such as fish populations recovering after the removal of fishing pressure, bird populations dropping sharply due to declines in the area and condition of grasslands, and forest-dwelling caribou at risk from fragmented forests (Federal, Provincial and Territorial Governments of Canada, 2010). The dramatic loss of sea ice in the Arctic has many current ecosystem impacts and is expected to trigger declines in ice-associated species such as polar bears (*see* Case Study 2). Important uncertainties and knowledge gaps remain, especially concerning the risks to vulnerable species and ecosystem types.

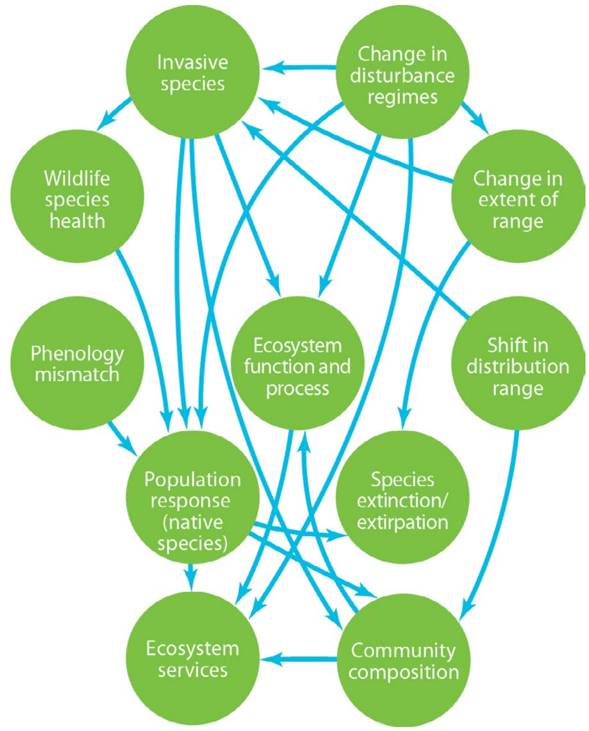

FIGURE 6: Illustration of the complex interactions among documented or projected responses of species and ecosystem processes to climate change. Such complexities make the outcome for any given species difficult to predict. Examples are provided throughout Section 3.

4. ADAPTATION AND THE ROLE OF PROTECTED AREAS

The conservation science literature contains a range of adaptation recommendations designed to reduce the effects of climate change on biodiversity (Figure 7; Heller and Zavaleta, 2009; Glick et al., 2011; Poiani et al., 2011; Oliver et al., 2012; Hounsell, 2012). These include improved institutional coordination, inclusion of spatial and temporal perspectives, and integrated coordination of climate change scenarios into planning and action related to ecosystem management. Natural responses of ecosystems to changes resulting from new environmental situations, called "autonomous adjustments" (SCBD, 2009) are generally considered insufficient to halt the loss of biodiversity and the ecosystem services it provides for people (Andrade Pérez et al., 2010). Planned adaptation aimed at maintaining or restoring biodiversity and ecosystem services, and thereby also helping people adapt to climate change, are becoming increasingly common worldwide (Andrade Pérez et al. 2010). Ecosystem-based adaptation (SCBD, 2009; Colls et al., 2009) focuses on building the resilience of local communities to climate change through ecosystem management that emphasizes the protection of biodiversity, restoration of ecosystem functions and the sustainable use of resources. Resilience refers to the capacity of an ecosystem and/or its cultural, social, and economic sub-systems to absorb or otherwise cope with change.

Conservation networks, with parks and other types of protected areas at their core, are key components of resilient socio-ecological systems because they protect ecosystem structure and function and provide connected habitats that offer the opportunity for organisms to respond to changing conditions. Protected area legislation provides the mechanisms to support "at-risk" species recovery, and protect cultural and social values (Fischer et al., 2009; Hounsell, 2012). Protected ecosystems provide important sites for research and monitoring (as relatively undisturbed benchmarks for measuring change, for example), as well as for engaging visitors and building public awareness about climate change impacts, opportunities, and solutions (NAWPA, 2012). Characteristics of successful network management programs include effective knowledge gathering and sharing, and provision for citizen engagement and participatory decision-making. Within Canada, both the Canadian Council on Ecological Areas (Lemieux et al., 2010) and the Canadian Parks Council (CPC, 2013) have examined the roles that the protection of well-functioning, well-managed ecosystems nested within sustainably managed landscapes and waterscapes play in maintaining resilience to climate change. In addition, the North American Commission for Environmental Cooperation has developed guidance for the design of resilient marine protected area networks in a changing climate (CEC, 2012). The following actions are considered core to adaption in support of ecosystem resilience (e.g. Mawdsley et al., 2009; Berteaux et al., 2010; Lemieux et al., 2010; Lindenmayer et al., 2010; CEC, 2012; Auzel et al., 2012).

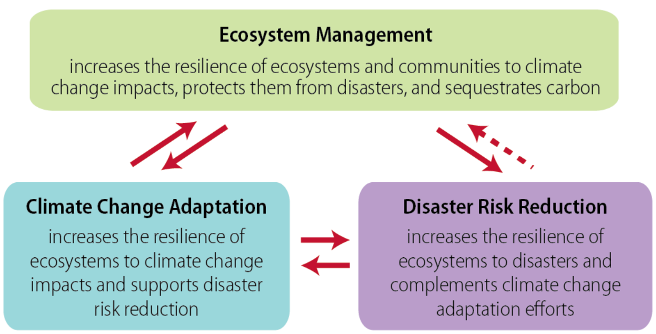

FIGURE 7: Illustration of the role of ecosystem management in increasing the resilience of natural systems and human societies, maximizing co-benefits of climate change mitigation, providing physical defence from climate-related disasters (e.g. coastal protection), and contributing to proactive adaptation and disaster management measures (*modified from UNEP, 2009*).

1. Protection of intact ecosystems and the diversity of species and processes that are part of them.
2. Connecting protected areas through sustainably managed landscapes and waterscapes.
3. Restoration of degraded ecosystems, and support of species recovery.
4. Maintaining or restoring natural disturbance regimes to reflect the natural range of variability characteristic for the ecosystem of interest.
5. Inclusion of conservation measures that protect and manage range limits.
6. Consideration of active management approaches, such as assisted migration, where appropriate.

This section explores these actions, provides examples of current Canadian research and initiatives, and describes their role in climate change adaptation. The effectiveness of adaptation actions requires knowledge gained through research and monitoring, along with traditional knowledge (e.g. Lalonde et al., 2012). It also requires careful planning based on shared understanding, engagement and support amongst people that enables them to assess and cope with change and take advantage of new opportunities (Chapin et al., 2009).

4.1 PROTECTING INTACT ECOSYSTEMS

Protected areas are critical for the conservation of biodiversity in periods of rapid environmental change. Protected areas provide habitat for native species and opportunities for autonomous adaptation, migration, and natural selection processes through maintenance of genetic diversity (Hannah et al., 2007; Hannah, 2009; Environment Canada, 2009; SCBD, 2009; Federal, Provincial and Territorial Governments of Canada, 2010). This in turn enhances species' capacity to respond to climate change impacts such as phenological mismatch and changing disturbance regimes (*see* Sections 3.1 and 3.4) with related societal benefits as a result (Dudley et al., 2010; NAWPA, 2012; CPC, 2013).

Recent protected area establishment in Canada includes a national network of connected Marine Protected Areas (MPAs) in Canada's three oceans and the Great Lakes (Government of Canada, 2011; ICES, 2011). The federal government and several provinces are beginning to incorporate climate change adaptation when planning expansions of their parks and protected areas. Examples include increasing the size of Nahanni National Park Reserve in 2009, the creation of the Nááts'ihch'oh National Park Reserve in 2012, and the establishment of the Gwaii Haanas National Marine Conservation Area Reserve and Haida Heritage Site next to the existing Gwaii Haanas National Park Reserve in 2010.

4.2 CONNECTING PROTECTED AREAS THROUGH SUSTAINABLY MANAGED LANDSCAPES AND WATERSCAPES

Species and ecosystems will respond to climate change through evolutionary processes, as well as through autonomous adjustments, such as migration within connected land/waterscapes to climatically and ecologically suitable areas. Conservation planning differentiates two general types of connectivity – corridors (strips of habitat connecting otherwise isolated patches) and stepping stones (series of small patches connecting otherwise isolated patches) (Baum et al., 2004). While the incorporation of information on changing climate suitability over time will be important for connectivity planning, corridors such as valleys, riparian edges and coastal areas will remain critical mechanisms for species migration into new habitats. Such corridors also allow gene flow and provide food, water and shelter during periods of natural scarcity, significant disturbances (e.g. large wild fires), or disruption by human activities (SCBD, 2009; Government of Canada, 2010; Lemieux et al., 2011).

Although few land use plans in Canada have historically addressed habitat connectivity, this is changing. For example, the province of Nova Scotia and the Town of Amherst are collaborating to protect the Chignecto Isthmus, the strip of land that connects Nova Scotia to the North American mainland, as a wilderness area while protecting Amherst's drinking water (Government of Nova Scotia, 2008, 2012). In Ontario, the Ministry of Natural Resources 50 Million Tree program has targeted areas to create corridors between core natural areas on the most densely populated, highly fragmented landscapes in Canada (G. Nielsen, Ontario Ministry of Natural Resources, personal communication).

Since many protected areas in Canada are surrounded by managed forests, forest management plans seek to protect many ecosystem goods and services including the maintenance of landscape connectivity (*see also* Chapter 3 – Natural Resources). In Ontario, 'Areas of Concern' are established within areas selected for timber harvest operations (MNR, 2004). These Areas of Concern contain one or more natural or cultural values that merit protection and are afforded special consideration in forest management and during harvest operations. These types of areas can serve as corridors and stepping stones.

Large-scale planning for connectivity is also recommended in the marine environment (CEC, 2012). Marine habitat connectivity can be particularly important for connecting critical places for different life stages (e.g. larval versus adult) as distributions of key species shift in response to climate change (CEC, 2012). The Oceans Action Plan (DFO, 2005) identifies five large-scale ocean management areas where integrated management planning is now taking place: Eastern Scotian Shelf, Gulf of St. Lawrence, Placentia Bay/Grand Banks, Beaufort Sea, and Pacific North Coast.

Both temporal and spatial connectivity need to be considered in order to maintain a suitable climate for a given species or ecosystem type over a defined period of time (Rose and Burton, 2009; Pellatt et al., 2012). When examining the results of bioclimate envelope modelling in the context of protected areas, a sense of what areas have the greatest potential to harbour species can be obtained by identifying areas that maintain a suitable climate envelope over relatively long periods of time (decades to century) (*see* Case Study 3; Hamann and Aitken, 2012; Pellatt et al., 2012).

4.3 RESTORING DEGRADED ECOSYSTEMS AND SUPPORTING SPECIES RECOVERY

4.3.1 ECOLOGICAL RESTORATION

Strategies aimed at maintaining ecological health or integrity by reducing the influence of non-climate stressors such as habitat fragmentation or pollution, or by reversing prior degradation through ecological restoration, are important for increasing resilience to climate change (Harris et al., 2006). Ecological restoration has been an important tool for decades and contributes to climate change adaptation by helping to prevent species extirpation and maintaining healthy ecosystems (e.g. Thorpe, 2012). Working landscape initiatives such as the Habitat Stewardship Program for Species at Risk (Environment Canada, 2012) include restoration strategies and techniques.

Protected-area agencies in Canada develop and apply ecological restoration techniques for individual species, biotic communities and whole ecosystems. These programs involve experimentation, modification, and adaptation, and create a culture with a capacity and willingness to adapt to change (Parks Canada, 2008). Guidelines have been developed for restoration techniques, now commonly used in many parks and other protected areas (Parks Canada Agency and the Canadian Parks Council, 2008). Examples include the restoration of grassland ecosystems in Grasslands National Park (Saskatchewan) and the restoration of riparian habitats and aquatic connectivity in La Mauricie National Park (Quebec) (Parks Canada, 2011a, b). More recent work includes explicit consideration of climate change adaptation and mitigation, relationships between the restoration of ecosystems in and around protected areas and ecological connectivity, visitor and public engagement and experience, human well-being, and the protection and provision of ecological goods and services (Keenleyside et al., 2012).

Programs to maintain or increase genetic diversity, such as seeding or sowing multiple species of native plants when converting marginal agricultural land to cover, and planting multiple tree species during forest landscape restoration, enhance the tolerance of species associations to change and can help build or maintain resilience to climate change (Thompson et al., 2009; Maestre et al., 2012) and other stressors. Efforts to restore the ecological integrity of Grasslands National Park have included the re-vegetation of previously cultivated fields with a mix of native grasses and wildflowers, and the re-introduction of disturbance regimes (e.g. the use of prescribed fire) needed for ecological integrity associated with grazing by bison (Parks Canada, 2011a).

CASE STUDY 3

TEMPORAL CONNECTIVITY OF GARRY OAK HABITATS IN PROTECTED AREAS OF WESTERN NORTH AMERICA

Garry Oak (*Quercus garryana*) ecosystems (Figure 8), found along the southern part of Canada's west coast, are rare and contain over one hundred species-at-risk. Examining how Garry Oak (*Quercus garryana*) could respond to climate change at scales relevant to protected area management assists in the design of monitoring programs and helps guide park managers and planners in selecting sites for protecting or restoring temporally connected areas that are climatically suitable for species of interest (Pellatt et al., 2012).

FIGURE 8: Garry oak on Saturna Island, Gulf Islands National Park Reserve.

Assessing climate change risks to Garry Oak (*Quercus garryana*) ecosystems involved evaluating the present level of protection of Garry Oak (*Quercus garryana*) and then forecasting how well currently protected areas in the Pacific Northwest encompass climatically suitable habitat under future climate scenarios. A down-scaled bioclimate envelope model was developed to identify areas projected to maintain climatic suitability over time. Scenarios were generated to examine temporally connected areas for Garry Oak (*Quercus garryana*) that persist throughout the 21st century, and the extent of overlap between these temporally connected regions and existing protected areas. Although climatically suitable Garry Oak (*Quercus garryana*) habitat is projected to marginally increase, mostly in the United States, this habitat will not be well represented in the World Conservation Union (IUCN) Class I-V Protected Areas. Of the protected area that currently encompasses suitable Garry Oak (*Quercus garryana*) habitat, models indicate that only 6.6 to 7.3% will be "temporally connected" between 2010 and 2099 (Figure 9; based on CGCM2-A2 model-scenario). Overlap between climatically suitable habitat for Garry Oak (*Quercus garryana*) (whether this suitable climate is currently present or not) and Class I-V protected areas in the 2010-2039 time period is approximately 40% using the same climate change scenario. Hence, Garry Oak (*Quercus garryana*) is poorly represented in temporally connected areas outside and inside protected areas, highlighting the need for public and private protected-area organizations to work cooperatively to maintain temporal connectivity in climatically suitable areas for the future of Garry Oak (*Quercus garryana*) ecosystems.

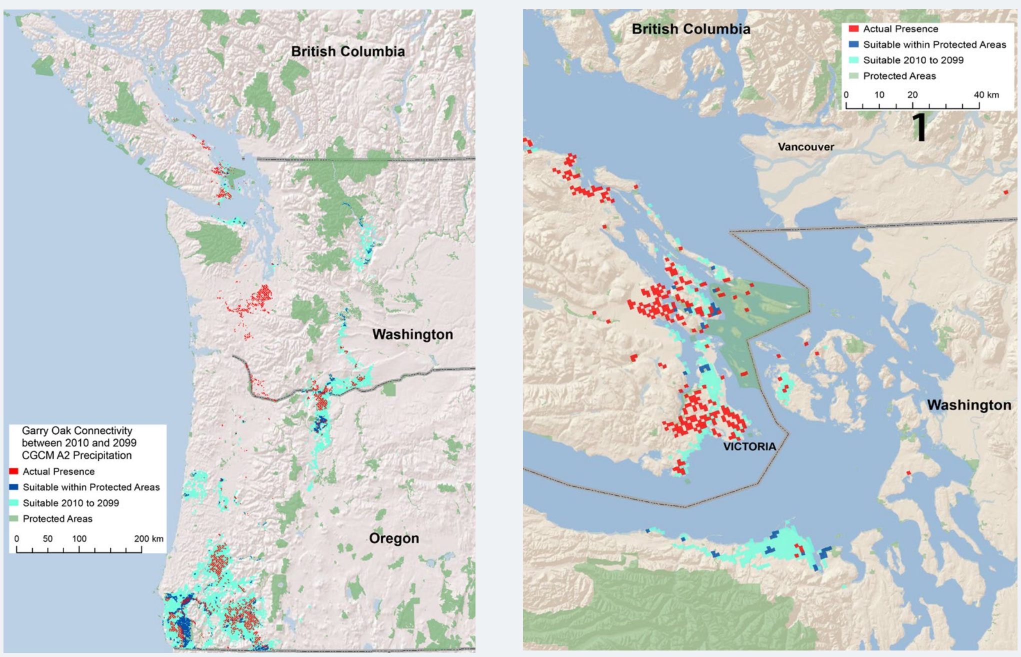

FIGURE 9: Climatically suitable habitat for Garry Oak *(Quercus garryana)* using scenario A2 (temporally connected) between 2010 and 2099. Green represents the location of protected areas. Light blue represents temporally connected Garry Oak habitat. Dark blue represents temporally connected Garry Oak habitat within existing protected areas. Red represents actual occurrence of Garry Oak.

Ongoing monitoring is helping to determine how the ecosystem is responding to these interventions and informing adaptive decision-making. This approach is also useful outside protected areas on the intervening land/waterscapes. For example, guidance for landowners, leaseholders and other land managers on Alberta's prairies (e.g. Saunders et al., 2006) encourages the use of native seed mixes in the restoration of rangelands to enhance the resilience of wildlife populations and the sustainability of agricultural operations.

As with other conservation measures, ecological restoration can also help to address the impacts of a changing climate on human well-being (*see* Chapter 7 – Human Health). For example, the protection and restoration of wetlands and natural drainage patterns helps to conserve water and is thus an important adaptation measure (Schindler and Bruce, 2012). The restoration of beaver populations that influence wetland hydrology and enhance water retention has led to significant increases in open water areas in Elk Island National Park (Alberta) even during exceptionally dry years (Hood and Bayley, 2008; Schindler and Bruce, 2012). Ecological restoration within and between protected areas helps to reduce landscape fragmentation and facilitates migration, gene flow, and other types of adaptation to changing conditions (*see* Case Study 4). Large-scale connectivity conservation initiatives such as the Yellowstone to Yukon Conservation Initiative (Graumlich and Francis, 2010), Algonquin to Adirondacks initiative (Algonquin to Adirondacks Collaborative, 2013), and the Wildlands Network's North American Wildways (The Wildlands Network, 2009; Dugelby, 2010) include ecological restoration as a key component of connectivity and climate change adaptation.

Climate change also creates significant challenges for ecosystem managers faced with establishing realistic targets for ecological restoration. Non-analogue, or "novel" ecosystems, may need to be accepted as management targets in some cases (Hobbs et al., 2009, 2011). Ecological restoration may thus be seen as part of a suite of interventions (i.e. "intervention ecology"; Hobbs et al., 2011) that take the form of manipulating the biotic and abiotic characteristics of the ecosystem, and can vary in intensity from deliberate non-intervention, through directed one-off interventions, to ongoing large-scale interventions. Incorporating climate change scenarios, including extreme events, into models that examine species response to climate (i.e. bioclimate envelope models, process models, observational studies, monitoring) as part of the analysis supporting ecological restoration, would permit improved assessment of whether: 1) natural recovery of a system is feasible, 2) intervention (e.g. assisted migration) would be necessary for species survival, 3) human engineering actions would be necessary (e.g. shoreline vulnerability to sea level change and coastal erosion, changes

CASE STUDY 4

A LANDSCAPE APPROACH TO ECOLOGICAL RESTORATION

(Source: Parks Canada, 2011a, b).

Ecological restoration principles are reflected in an initiative to restore landscape connectivity within the Long Point World Biosphere Reserve, which includes the Long Point National Wildlife Area along the northern shore of Lake Erie in Ontario (Figure 10). Conservation science and state-of-the-art information management technology were used to identify Carolinian core natural areas and other significant natural areas, as well as potential habitat corridors to link the natural areas together. This interconnected system of habitat cores and corridors was designed to facilitate dispersal of plants and animals to more favourable habitats and conserve biodiversity in the face of a changing climate. Conservation partners, which include the private sector, have supported efforts to plant over 4.5 million native trees and shrubs, and use restoration techniques to mimic features of old-growth Carolinian forests. This work is helping to create corridors and enhance ecosystem resilience and adaptive capacity throughout the Biosphere Reserve.

FIGURE 10: A pit and mound restoration technique was used to reproduce the characteristic pattern of old-growth forests created by the decay of fallen trees. The pits have provided breeding areas for amphibians and insects, feeding and drinking holes for birds and mammals, and have contributed to ground water recharge. The mounds, with well-drained and oxygenated soils, have allowed for rapid growth of Red and White Oaks, among other tree species (*Photo courtesy of B. Craig).*

in river hydrology and surface runoff; Galatowitsch, 2012), and 4) a novel ecosystem might need to be accepted (Hobbs et al., 2011). Given the uncertainties associated with the management of complex ecological processes, an adaptive management approach that regularly re-visits objectives and decisions, and adjusts them as knowledge advances, is particularly important for ecological restoration and related interventions in the context of climate change (Keenleyside et al., 2012).

4.3.2 ASSISTED MIGRATION

As projected changes in climate over the next century may exceed the natural capacity of many populations to adapt through migration or other responses (Loarie et al., 2009), human-mediated transport of selected species to more favourable climatic habitat may be a management option (Eskelin et al., 2011; Pedlar et al., 2012; *see also* Chapter 3 – Natural Resources). In addition to land management initiatives that increase connected habitat for migration, three different types of assisted migration (defined as human-assisted movement of species) have been identified (Ste-Marie et al., 2011):

1. *Assisted population migration:* The human-assisted movement of populations (with different genetic makeup) of a given species within that species' current range (i.e. where it would naturally spread).
2. *Assisted range expansion:* The human-assisted movement of a given species to areas just outside its current range, assisting or mimicking how it would naturally spread.
3. *Assisted long-distance migration:* The human assisted movement of a given species to areas far outside its current range (beyond where it would naturally spread).

Assisted population migration and assisted range expansion are currently used in Canada. For example, in Quebec, climate sensitive seed transfer models are used to identify sites (within a species current range) where seedlings produced from seeds grown in seed orchards can be planted for the best chances of survival and growth to maturity (Ste. Marie et al., 2011). Assisted range expansion is employed in British Columbia, where most tree species in most regions can be planted 200 m higher in elevation, and Western Larch can be planted slightly outside its current range (O'Neill et al., 2008), and in Alberta where seed zone limits have been extended by up to 2º north latitude and by up to 200 m upward in elevation (Ste-Marie et al., 2011). Assisted long-distance migration is considered in cases where the species is threatened or geographical/physical barriers prevent natural migration. This type of assisted migration is riskier than the other two techniques because it involves the introduction of new genetic stock that may damage other species and the ecosystem into which it is introduced. No long-distance management programs are currently planned by Canadian forestry agencies (Ste-Marie et al., 2011).

There are a range of views on the practicality and appropriateness of using assisted migration as an adaption tool (Riccardi and Simerloff, 2008; St. Marie et al., 2011; Winder et al., 2011; Beardmore and Winder, 2011; Aubin et al., 2011; Pedlar et al., 2011, 2012; Larson and Palmer, 2013). A thorough understanding of the risks is essential prior to this type of work. Operationally, it requires high resolution climate projections, genecological information (the gene frequency of a species in relation to its population distribution within a particular environment), information on traits that respond to selective pressures of climate variables, and models that predict appropriate seed sources for planting (St. Clair and Howe, 2007; O'Neill et al., 2008; Rehfeldt and Jaquish, 2010; Eskelin et al., 2011).

Although assisted migration could minimize species loss under rapid climate change, it could also pose a significant risk of disrupting existing communities (McLachlan et al., 2007). While great effort has been spent studying invasive species, it is difficult to predict which species would become pests. Furthermore, the lag between introduction and population explosion in exotic invaders can be decades long, suggesting that efforts to monitor relocated species for negative ecological consequences could become impractical (McLachlan et al., 2007). While some of the risks of assisted migration can be minimized through planning, monitoring, adaptive management and regulation (Mawdsley et al., 2009), support for and uptake of assisted migration techniques will vary according to the goals and objectives in protected areas and on the intervening landscapes and waterscapes.

4.4 PLANNING FOR ADAPTATION

Many conservation initiatives in North America are increasingly incorporating information about climate change. In Canada, ongoing enhancements of protected area networks make a significant contribution to reducing the effects of climate change and providing opportunities for adaptation. Examples include the Natural Areas System Plan being developed by the government of Newfoundland and Labrador (Government of Newfoundland and Labrador, 2011), two new parks established in Manitoba in 2010 to protect the winter habitat of the Qamanirjuaq barren-ground caribou herd (Government of Manitoba, a, b, [n.d.]), and the 'Recommended Peel Regional Watershed Land Use Plan' in northeast Yukon. The latter is the outcome of a regional planning initiative designed to help ensure wilderness, wildlife and their habitat, cultural resources, and waters are maintained under rapid climate change (Lemieux et al., 2010). Produced as part of the implementation of Chapter 11 of the

Final Agreements for the Na-cho Nyak Dun, Tr'ondëk Hwëch'in, and Vuntut Gwitchin First Nations, the plan explicitly links climate change impacts and the need for protected areas that represent large, intact ecosystems (Peel Watershed Planning Commission, 2010).

Another strategy for conserving regional biodiversity in a dynamic climate is to ensure that conservation planning encompasses the full spectrum of physical settings; defined by elevation, geology and many other factors (Beier and Brost, 2010). As contrasting physical settings maintain distinctive ecological communities in a variety of climates (Rosenzweig, 1995), conserving representative examples of these settings should help protect biodiversity under both current and future climates (Beier and Brost, 2010). Game et al. (2011) found that protected areas identified using physical diversity captured over 90% of the diversity in vegetation communities.

4.5 BUILDING KNOWLEDGE TO SUPPORT PLANNING

Integrated climate change biodiversity assessments are one important means of building the necessary knowledge to identify and evaluate preparedness and intervention options, and to enhance adaptation to climate change. Pellatt et al. (2007) undertook such a study using paleoecology, dendroecology, spatial analysis, and bioclimate envelope modelling to develop a better understanding of the future of Garry Oak and Coastal Douglas-Fir ecosystems (*see* Case Study 3). This work has since been expanded to include the establishment of a long-term restoration experiment that monitors ecosystem response to environmental change, including the incorporation of prescribed fire and the exclusion of large herbivores from the ecosystem. Empirical and observational studies are required to assist the interpretation of climate suitability scenarios generated by climate models (Hamann and Wang, 2006; Hamann and Aitken, 2012).

Research is similarly informing management decisions in Kouchibouguac National Park (New Brunswick) where aerial photos, surveys and fieldwork are being used to document coastal zone change over the past few decades and to identify areas potentially requiring more protection in the future (Parks Canada, 2010). Likewise, British Columbia Parks is examining the sensitivity of the coastline to sea level rise to inform management plans for new protected areas along the north and central coast (Province of British Columbia, 2013).

Ecological monitoring is another foundation for the knowledge needed to inform climate change adaptation (Hannah et al., 2002). For example, arctic-alpine vegetation is being monitored in Quebec's parks with alpine zones (e.g. Parc national de la Gaspésie, des Hautes-Gorges-de-la-Rivière-Malbaie and des Grands-Jardins; Société des établissements de plein air du Québec, 2012) as indicators of climate-related change that will affect caribou (a species at risk) and other species that rely on habitats comprised of these plant species. Similar monitoring programs that use both land-based and remote (including space-based) technologies serve as tools to report on the state of ecological integrity in National Parks across Canada (McLennan and Zorn, 2005; Parks Canada, 2011c).

Alongside other types of scientific knowledge, Aboriginal knowledge and community experiential knowledge should contribute more broadly in informing climate change adaptation planning and conservation decision-making (CPC, 2013). This knowledge provides valuable information on historic and current ecosystem conditions and human-ecological interactions based on hundreds or even thousands of years of experience (Waithaka, 2010). For instance, Torngat Mountains National Park (Newfoundland and Labrador) is involved in research projects to study key Inuit food sources such as berries and ringed seals, and to establish baseline data to assess the effect of future climate changes on these important species (CPC, 2013).

4.6 ENGAGING COMMUNITIES IN ADAPTATION

Collaboration is fundamental to ensuring that adaptation actions related to other sectors (e.g. built infrastructure) do not negatively impact biodiversity or the ability of ecosystems to respond to change, and that actions aimed at helping biodiversity to respond to climate change also bring societal benefits (Andrade Pérez et al., 2010). Key players include conservation organizations, park visitors, local community groups, Aboriginal communities, and industry. Successful engagement leads to more responsible decision-making and promotes sustainable approaches to natural resource stewardship (NAWPA, 2012).

There are many examples of effective engagement and collaboration in Canada, ranging from citizen-based monitoring programs (*see* Case Study 5) to overseeing policy development at municipal, provincial/territorial and national levels. These include a climate change vulnerability assessment and adaptation options pilot study for the Clay Belt ecosystem in northeastern Ontario, where stakeholders and partners were encouraged to participate from the outset and were apprised of the results of completed work and overall study progress (Lalonde et al., 2012). An example of inter-governmental collaboration is the Manitoba-Ontario Interprovincial Wilderness Area that encompasses more than 9400 km^2 of boreal forest and includes core protected areas in provincial parks and conservation reserves (CPC, 2013). First Nations communities in both provinces are seeking to create a 30 000 km^2 network of protected areas and managed landscapes on ancestral lands.

CASE STUDY 5

CITIZEN-BASED MONITORING PROGRAMS

Monitoring provides critical data for adaptation planning, and can be undertaken at a range of scales involving different groups. Planners and the scientific community are increasingly drawing on information collected by community groups. Two examples described here relate to the American Pika and to marine fish and invertebrates.

Sensitive species such as American Pika (*Ochotona princeps*; Figure 11) can be used as an indicator species for detecting the impact of climate change in rocky mountainous ecosystems. In Banff and Kootenay National Parks (Alberta and British Columbia), collaboration between Parks Canada, the University of Alberta, and the Bow Valley Naturalists Society has allowed for engagement of local community members in ecological monitoring activities, while maintaining quality control. Surveyors (2 to 4 people) searched a given block of pika habitat (e.g. talus pile) for hay piles and pikas within 30 m of the talus edge. The locations of hay pile clusters that actively supported or did not support pikas were recorded with a GPS (Global Positioning System), as were actual pika observations (Timmins and Whittington, 2011). This type of monitoring will assist management in determining the impact of climate change on pika habitat and inform management decisions.

FIGURE 11: Pika (*Ochotona pinceps*) in a talus slope in Banff National Park.

Large-scale, long-term data on species distribution and abundance is rare for marine organisms, especially those that are not subject to monitoring for fisheries purposes. Since 1998, recreational divers in BC have been partaking in the Reef Environmental Education Foundation's (REEF) volunteer fish and invertebrate monitoring program. Participants are trained to identify target species and implement a simple roving diver survey method. More than 3700 volunteer surveys have been carried out along the British Columbia coastline through this program, representing more than 2800 hours of underwater observations at more than 300 sites, and documenting the abundance of nearly 150 fish species and 50 invertebrate species. The resulting 15-year-long time-series documents abundance trends (*see* Figure 12) for a wide range of species. Results also help to establish species range limits, as well as changes in species distribution and shifts in community assemblages over time. The program was extended to include eastern Canada in 2012.

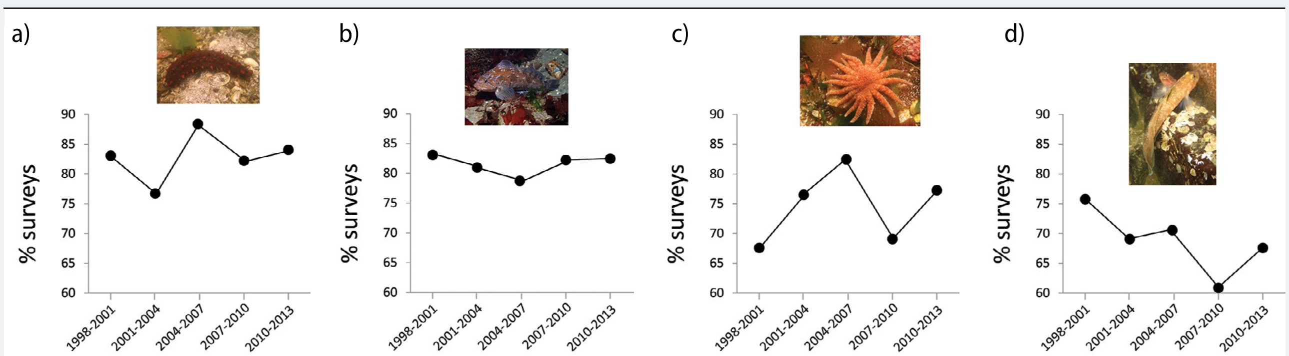

FIGURE 12: Abundance trends of four of the marine species most commonly encountered by recreational divers taking part in REEF surveys in British Columbia. **a)** California sea cucumber (*Parastichopus californicus*), **b)** kelp greenling (*Hexagrammos decagrammus)*, **c)** sunflower seastar (*Pycnopodia helianthoides*), and **d)** blackeye goby (*Rhinogobiops nicholsii*). Abundance is expressed as the proportion of surveys in which each species was observed. The number of surveys generating each point varied from 83 (invertebrates, 1998-2001) to 1074 (fishes, 2010-2013) *(Photos courtesy of I.M. Côté. Data derived from www.reef.org)*.

5. CONCLUSIONS

Evidence that Canada's biodiversity is under increasing pressure from climate change continues to grow. Changes in the timing of life history events, species distribution ranges and wildlife health are already evident and are predicted to increase. Biodiversity underpins the well-being and prosperity of Canadians through the provision of ecosystem services. Strategies that help to maintain and restore biodiversity not only help ecosystems respond to climate-related changes, but also enhance ecological, social and economic resilience.

The extent and pace of climate change is creating a new ecological context in which natural resource managers are increasingly considering more interventionist approaches to biodiversity conservation (Glick et al., 2011; Poiani et al., 2011). Practices such as assisted migration are being considered alongside ecological restoration and other interventions in an effort to manage change (Glick et al., 2011).

The relationship between climate change impacts on biodiversity discussed in Section 3 of this chapter, and the adaptation strategies that are being used to help biodiversity respond to change, is illustrated in Figure 13. Actions aimed at protecting, connecting or restoring networks of well-functioning ecosystems in protected area networks, along with conservation-focused habitat stewardship on private lands and waters, and sustainable land and water use practices (e.g. sustainable forestry, agriculture, and fisheries) are enhancing the resilience of Canada's natural capital. These actions are informed by new knowledge about climate-driven changes in ecosystems, by the integration of that knowledge into conservation planning, and by the development of new partnerships and collaborative processes that include broad-based engagement of Canadians.

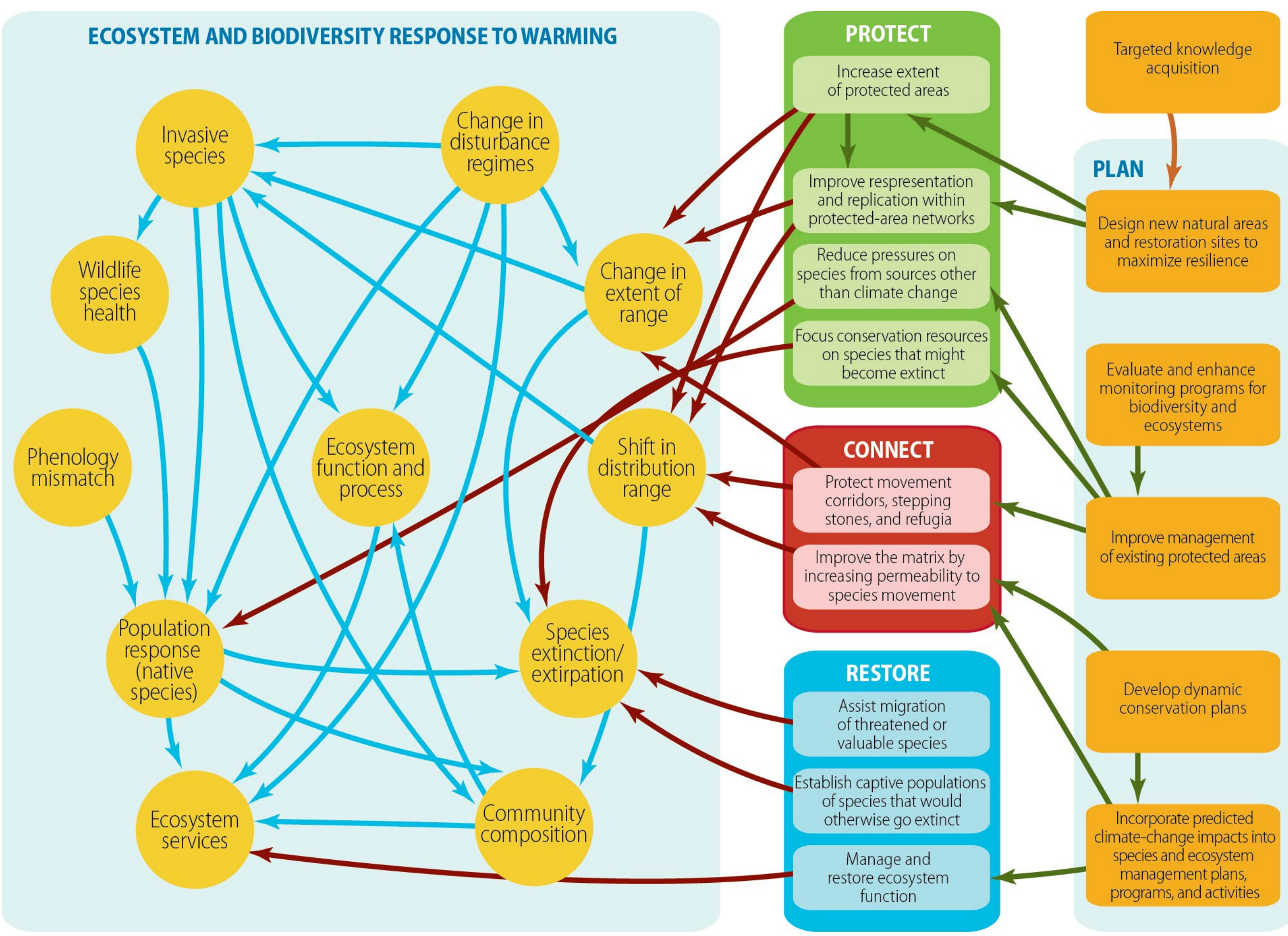

FIGURE 13: Linkages among elements of a conservation-based strategy for biodiversity adaptation to climate change and their potential effect on individual ecosystem response.

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