Biodiversity Conservation in Canada: From Theory to Practice

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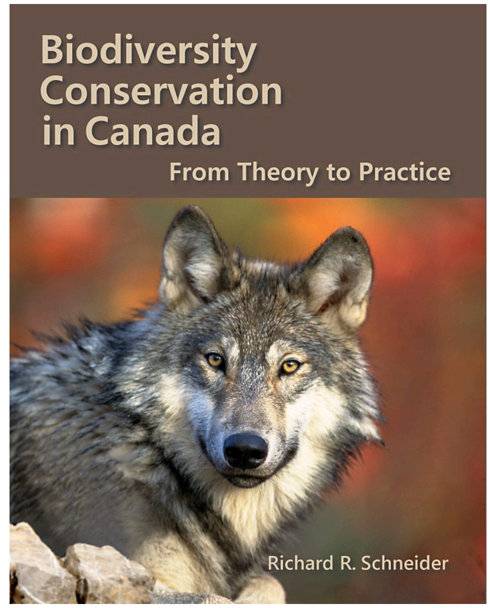

Dedication

To my mentors, Jacob Schneider and Peter Yodzis, and to my family

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Preface

Conservation is often portrayed as an applied science—a corpus of knowledge about how ecological systems function, how they are threatened, and how they can be maintained. Conservation is also a form of management. It entails working with people to achieve desired ecological outcomes, grappling with conflicting land-use objectives, and making optimal use of available conservation resources. The aim of this book is to build a bridge between these two perspectives, linking theory with practice.

The challenge in wading into the practical aspects of conservation is that much depends on local circumstances. The types of threats matter; the existing laws and policies matter; institutions matter; the values and concerns of local people matter; history matters, and so on. In short, how conservation is done depends on where it is done. Most conservation texts address this issue by incorporating case studies from different regions. Readers are left to figure out for themselves how all the pieces fit together and how conservation is actually practiced in their region.

In this book, I take a different approach. The entire narrative is structured around one specific region: Canada. This permits an integrated treatment, where conservation theory is presented in the context of the social and institutional framework responsible for its implementation. The result is a synthesis tailored to the needs of conservation students and practitioners in Canada.

For undergraduate students studying conservation, this text will serve as an introduction to the field, providing a broad overview of both the scientific and social dimensions of conservation. For graduate students who have already acquired a solid foundation in ecological theory, this book will serve as a gateway to conservation practice, showing how the scientific "tools of the trade" are applied in real-world settings. For readers with a professional or personal interest in conservation, this book will provide an accessible description and guide to how conservation is currently practiced in Canada.

Given the book's applied focus, special attention is given to issues that are the subject of debate or controversy. Conservation practitioners will invariably encounter these issues and should be prepared to deal with them. Moreover, these debates provide valuable insights into the practical aspects of conservation. Some of the issues explored in the text include:

- How do we determine how much conservation is enough—a ubiquitous question that arises in the context of protecting habitat, limiting industrial activities, and many other applications
- What do we mean by "natural"?
- Is the aim of conservation to achieve the most good for the most species, or is it acceptable to prioritize some species over others?
- Are conservation practitioners dispassionate scientific advisors or biodiversity advocates?
- What does it mean to maintain biodiversity when ecosystems are fundamentally changing because of global warming?
- Does the advent of "new conservation," centred on the delivery of ecosystem services, support or hin-

der the conservation of biodiversity?
- What is the difference between science and Indigenous traditional ecological knowledge, and what are their respective roles in decision making?

Chapter 1 introduces fundamental concepts and provides a conservation framework that subsequent chapters build on. The next four chapters provide the social and scientific context of conservation, setting the stage for the applied chapters that follow. In these initial chapters, we learn the "what" and "why" of conservation. We are also introduced to the factors that constrain it.

Chapters 6–8 are devoted to the practice of conservation at both the species level and the ecosystem level. In these chapters, we learn the "how" of conservation and acquire an understanding of the relationship between theory and practice. We also gain insight into the role of conservation practitioners in the overall enterprise of conservation.

Chapter 9 presents an overview of the ecological changes expected as a result of climate change and what these changes mean for conservation. Specific consideration is given to the adjustments that need to be made to conservation objectives and conservation practices.

Chapter 10 focuses on decision making and planning. This topic is often overlooked by conservation students, who may be more interested in the ecological aspects of conservation. Nonetheless, it is central to the practice of conservation. Not much happens without a policy, strategy, or implementation plan to encapsulate decisions about priorities, limits, and choices among management alternatives. Knowing how to make effective decisions is an essential skill for conservation practitioners.

Chapter 11 presents a suite of integrated case studies—both successes and failures. This chapter revisits the main conservation themes of the earlier chapters from a practical perspective. Here we see how all the pieces fit together. The case studies are designed to illustrate the complexities of real-world conservation and to provide additional insight into how conservation theory is translated into practice.

The final chapter examines applied conservation from the perspective of effectiveness. We look back over what we have learned, highlighting the factors that are typically associated with successful conservation outcomes. We also discuss the training and tools needed for maximizing effectiveness at the personal level.

My hope is that readers will come away from this book with a solid understanding of the "big picture" of conservation in Canada. There is, of course, much more to learn, both in terms of ecological theory and management techniques. But this book will provide a sound foundation to build on. Moreover, readers will have a clear sense of what it is to be a conservation practitioner and how to be effective in this role.

Digital Edition

The digital edition of *Biodiversity Conservation in Canada: From Theory to Practice* presents a fully revised version of the original print book. Factual information that has changed since the original publication, including graphs and tables, has been updated with the most recent data available as of 2023. The text has also been revised and a new section, on citizen science, has been added.

The digital version of the book is published as an Open Text, under a Creative Commons licence. The intent is to make the book as widely available by providing access through the Internet and by removing cost as a barrier. In addition, educators and others are welcome to use the illustrations, tables, and text excerpts to support their work in whichever manner they please.

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This book was commissioned by Stan Boutin, who, at the time of writing, held the Alberta Biodiversity Conservation Chair at the University of Alberta. This project began as a legacy initiative, building on over 20 years of collaborative efforts involving myself and Stan and a network of colleagues. I thank Stan for all the support he has provided to me over the years, and for providing the seed funding, encouragement, and advice that made this book possible.

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CHAPTER I

AN INTRODUCTION TO CONSERVATION

An Introduction to Conservation

Defining Biodiversity

We will begin our exploration of conservation with a high-level overview, starting with the components of biodiversity—the targets of our conservation efforts. We will then sketch out, in broad strokes, the biocentric and social models of conservation practice, establishing a foundation that subsequent chapters will build on.

Biodiversity refers to the variety of life in all its forms and at all levels of organization. Three levels of organization are usually distinguished: species, genes, and ecosystems. We will examine each of these levels in turn, emphasizing foundational concepts. In later chapters, we will revisit these concepts from the perspective of applied conservation.

Species Diversity

A species is defined as a group of organisms capable of interbreeding under natural conditions. However, this definition is difficult to apply in practice because it requires information about reproduction patterns that is generally unavailable. Therefore, taxonomists commonly differentiate species on the basis of physical and functional characteristics and, more recently, DNA barcodes (Coissac et al. 2016).

Researchers estimate that the total number of species on earth is somewhere between 2–8 million, not including bacteria or viruses (Costello et al. 2013). Species diversity is greatest in tropical regions and declines as one moves poleward (Willig et al. 2003). For example, the Amazonian lowlands contain an estimated 16,000 tree species, whereas there are less than 200 native tree species across all of Canada (Ter Steege et al. 2013).

In total, Canada has approximately 80,000 known species (excluding bacteria and viruses), and there are many yet to be identified (CESCC 2022). Vertebrates and vascular plants have been well studied but the vast majority of species are simpler life forms (Fig. 1.1), and only a fraction of these species have been catalogued. The more we look, the more we find.

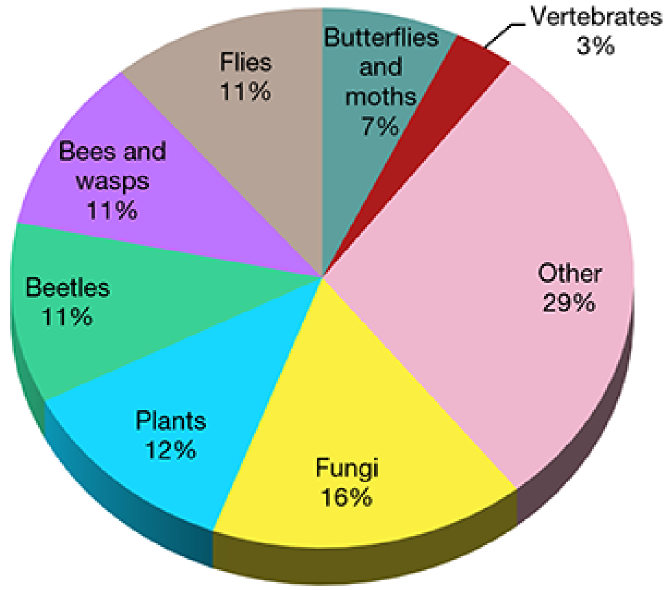


Fig. 1.1. Canada has approximately 80,000 known species, and insects comprise the largest group. Vertebrates, which garner most conservation attention, only account for 3% of species. Source: CESCC 2022.

A common measure of species diversity is **species richness**, which is the number of species in a given area. There are also more refined measures of diversity that take spatial scale into account (Jost 2007). These include **alpha diversity**, which is the number of species in a specific location; **beta diversity**, which describes how species composition changes across space; and **gamma diversity**, which describes the overall number of species across a broad region (Fig. 1.2). In real landscapes, these measures are often correlated.

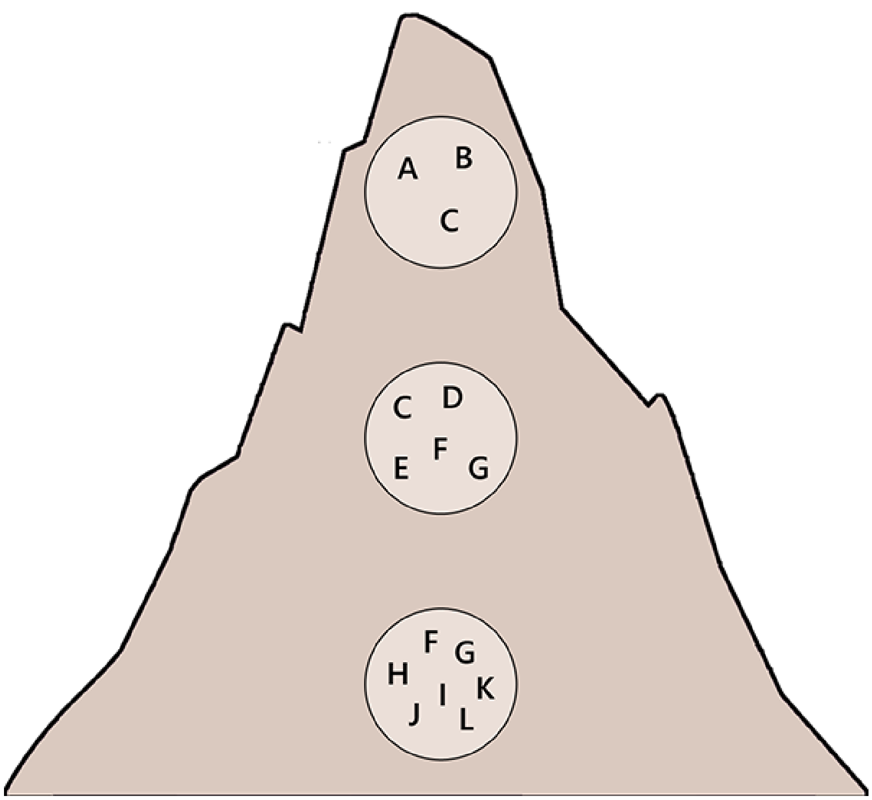


Fig. 1.2. A conceptual illustration of diversity patterns on a mountainside. The circles represent study sites, and the letters represent the individual species present. In this example, alpha diversity—the number of species within each site—decreases with altitude. There is little overlap in species composition among sites; therefore, beta diversity is high. Gamma diversity is the total number of species on the mountain (i.e., 12).

Of course, it is not only the number of species that matters but the diversity in traits and life strategies within the community. Important aspects of this diversity include:

- Functional role (e.g., predator, herbivore, decomposer, etc.)

- Habitat use (e.g., habitat specialist or generalist)
- Life history strategy (e.g., short lifespan, many offspring at a time, and minimal parental care vs. long lifespan, few offspring at a time, and high parental investment)
- Behavioural traits (e.g., herding vs. solitary; migratory vs. stationary)

Another attribute of relevance to conservation is whether a species is common or rare. There are three forms of **rarity** to be considered. First, some species are rare because they normally exist at very low population densities (e.g., the wolverine). Second, some species are associated with uncommon habitat types (e.g., patches of recently burned forest). Third, some species are rare because they have a very localized range (e.g., the Vancouver Island marmot).

The term **endemic** is often used to describe species with a restricted range. For example, we would say that the Vancouver Island marmot is endemic to Vancouver Island. While this usage of the term is correct, it should be noted that range size is not actually part of the definition. Endemic simply means that a species is found only in one geographic location. So, for example, the muskrat is endemic to North America.

Genetic Diversity

Genetic diversity arises from spontaneous DNA mutations that are passed down from parent to offspring. These mutations are rare, but they accumulate within a species over evolutionary time. The full complement of genetic variants that exists within a species is referred to as its **gene pool**. The variant forms of individual genes are referred to as **alleles**.

Through natural selection, beneficial mutations tend to become widely distributed among individuals, whereas deleterious mutations tend to be weeded out. However, because environmental conditions fluctuate over time, natural selection is faced with a moving target. In the face of changing conditions, no single set of genetic variants will be universally optimum (Reed et al. 2011). Instead, species are best served by having a diverse gene pool containing genetic variants that are adaptive under different conditions (Kremer et al. 2012). This allows species to adjust to environmental changes much faster than if they had to wait for new adaptive mutations to arise.

Another aspect of genetic diversity is spatial structure. Many species are divided into distinct **populations**—groups of individuals that live in a particular geographical area and normally breed with one another. Populations that reside in separate regions may experience different environmental conditions and different selective pressures (Fraser et al. 2011). This can result in localized genetic differences, especially if the rate of **gene flow** (i.e., reproductive mixing) among populations is low because of physical barriers or distance (Savolainen et al. 2007). Given enough time and isolation, sufficient differences can accumulate within populations to result in the formation of new species.

In Chapter 6, we will examine genetic diversity as it relates to extinction dynamics and the designation of species at risk. We will also discuss methods for maintaining genetic diversity. In Chapter 9, we will revisit genetics in the context of adapting to global warming.

Ecosystem Diversity

An **ecosystem** is a group of interacting organisms and the physical environment they inhabit. The living members of an ecosystem are referred to as a **community**. Unlike species and genes, ecosystems are not discrete entities. Although some distinct boundaries do exist, such as between a lakeshore and forest, most ecosystems change gradually across space in response to environmental gradients.

For example, hiking up a mountain trail, we find that the mix of tree species gradually changes with altitude, and at the highest elevations, trees become stunted and eventually give way to open meadows. These changes are driven by a climatic gradient: higher elevations are associated with lower mean temperature, higher moisture levels, and shorter growing season. Because the climatic changes are gradual, so too are the ecological transitions.

The factors that generate ecological patterns are scale-dependent (Willis and Whittaker 2002). At broad scales (like provinces), regional climate and large physical features are most important. At intermediate scales, soils, topography, and large-scale disturbance events (e.g., fire and insect outbreaks) are the major drivers. At the local scale, biotic interactions, small-scale disturbances, and microclimate have an overriding effect.

The ecosystem boundaries we define are largely artificial constructs developed for specific management purposes. For example, when the focus is on aquatic biodiversity, ecosystems may be delineated using watershed boundaries. The classification system most relevant for terrestrial conservation planning is the *National Ecological Framework for Canada* (Fig. 1.3), developed by the federal and provincial governments (Marshall et al. 1999). This classification delineates ecosystems on the basis of climate, topography, geology, soils, hydrology, vegetation, and fauna. Ecosystems are defined hierarchically, with three main divisions: ecozones (n=15), ecoregions (n=194) and ecodistricts (n=1,021). Some provinces have developed classifications that extend this hierarchy to even finer scales.

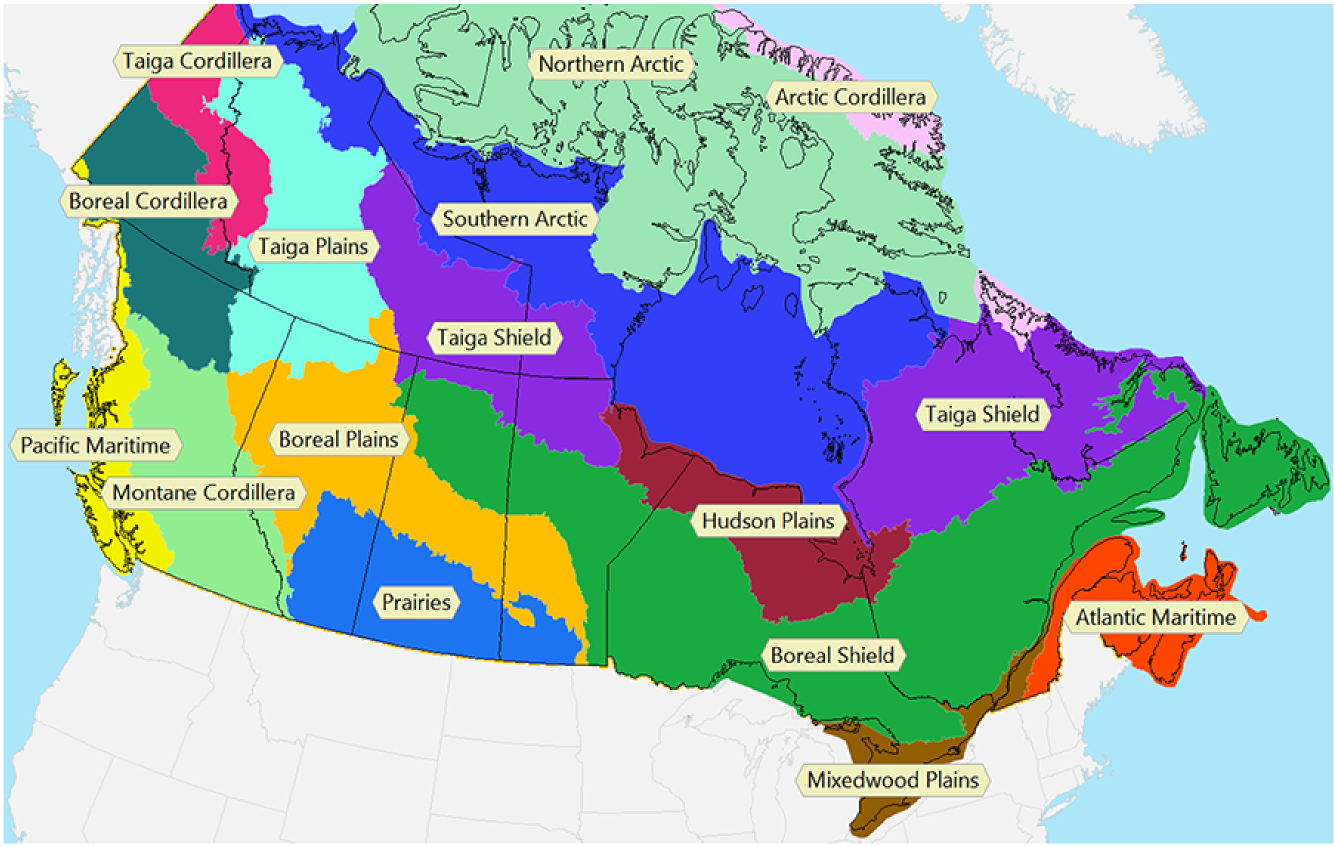


Fig. 1.3. The National Ecological Framework for Canada divides the country into 15 ecozones. Each ecozone is further subdivided into ecoregions and ecodistricts (not shown).

Individual ecosystems are described in terms of three main attributes: **composition**, **structure**, and **function**. Composition refers to the individual elements that comprise the system, especially the species that are normally present. Structure refers to how these elements are arranged in three-dimensional space. For example, consider the horizontal and vertical structure of a forest. Structure also includes the relative abundance of species. Some ecosystems are dominated by just a few species, whereas others have a more even distribution of species. Function refers to the various processes that occur within the system, both biotic (e.g., competition) and abiotic (e.g., nutrient cycling).

A process of particular relevance to conservation is disturbance. Natural disturbances (fires, insect outbreaks, storms, droughts, etc.) are a normal and necessary part of ecological systems. They reset **succession**, resulting in much greater ecological complexity than would otherwise exist. In contrast, **anthropogenic** disturbances (i.e., of human origin), especially those with no natural analog, can disrupt ecosystems and cause species to decline (see Chapter 5).

The Biocentric and Social Models of Conservation

Having clarified what biodiversity is, we can now consider what it means to conserve it. The *Oxford English Dictionary* defines conservation as the preservation, protection, or restoration of the natural environment and wildlife. From this definition we can infer three fundamental tasks for conservation practitioners: (1) determine what the natural state is; (2) determine how the natural state is being perturbed (i.e., identify threats); and (3) develop and implement measures to mitigate the disturbances and, if necessary, restore the system. We will refer to this as the **biocentric model** of conservation (Fig. 1.4). This model portrays conservation as a technical problem-solving process—conservation as applied science.

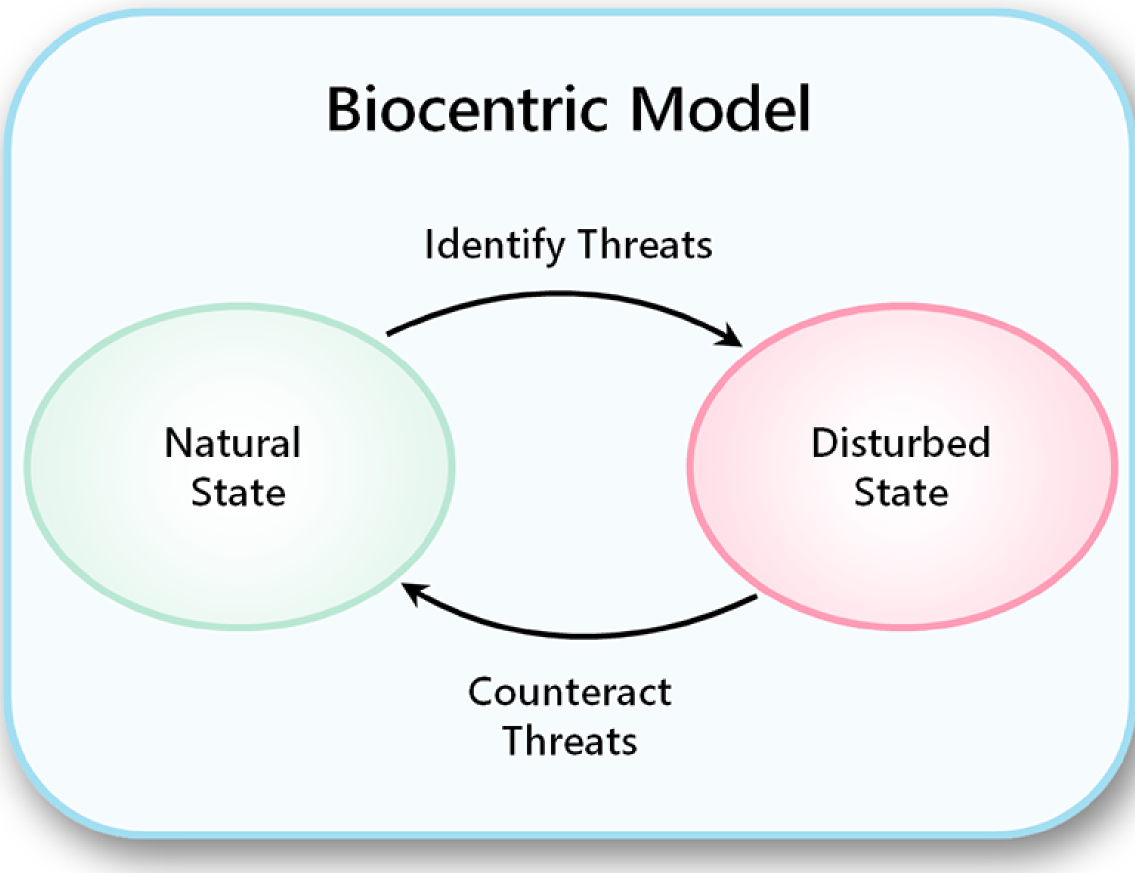


Fig. 1.4. The biocentric model of conservation is focused on maintaining (or restoring) the natural state of biodiversity. Conservation is seen as a technical problem-solving enterprise within the domain of science involving the identification and mitigation of threats.

The biocentric model of conservation is prominent in the conservation literature. However, it is incomplete and does not reflect the way that conservation works in practice. What is missing is the social dimension. Canadians value biodiversity, but it is not the only thing we care about. The ecological systems that support wild species also provide us with food, raw materials, opportunities for recreation, and other benefits. The problem this presents is that ecological systems have a finite capacity and cannot meet all the demands placed on them. In short, we cannot have everything we want. Consequently, **trade-offs** among competing values are a central feature of conservation (McShane et al. 2011). This leads us to the **social model** of conservation, which is centred on managing human activities (Fig. 1.5).

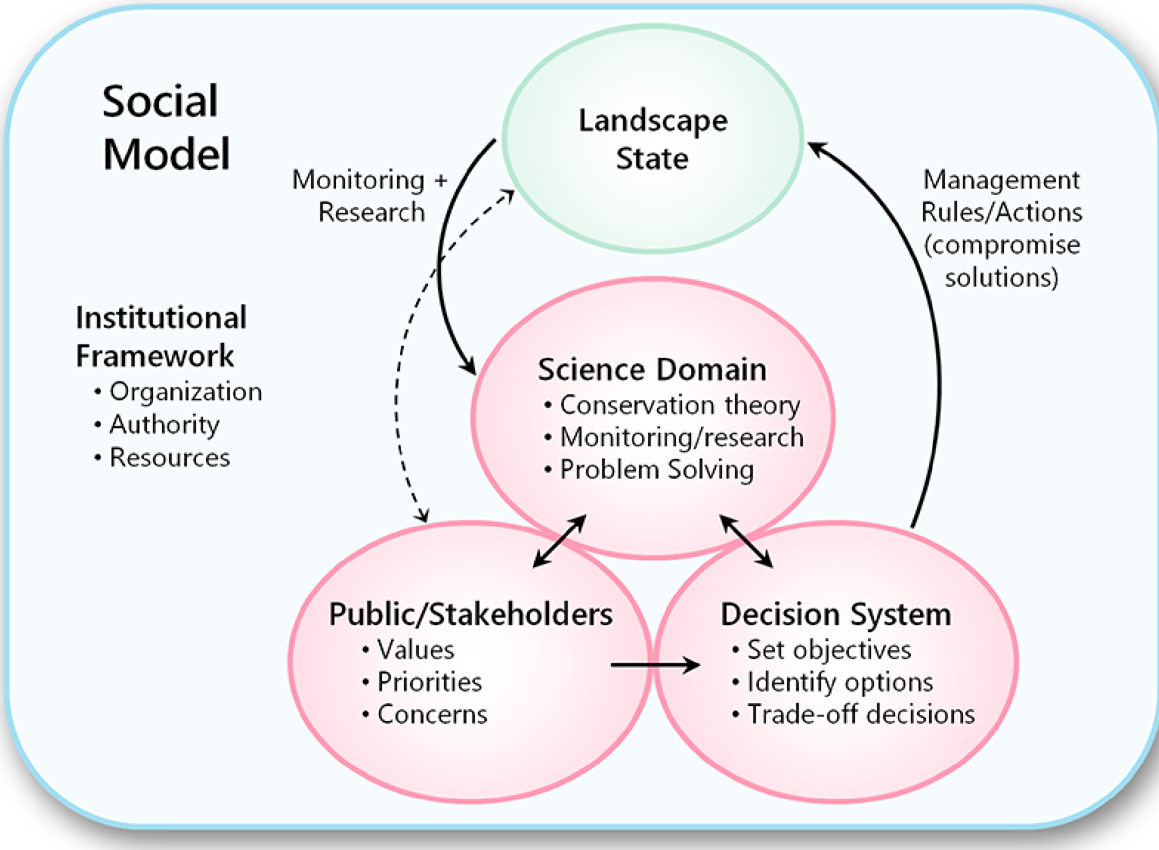


Fig. 1.5. The social model of conservation is concerned with maintaining biodiversity but takes other values into account as well. It includes a social values component, a science component, and a decision-making component. These three core elements exist within an institutional framework that provides organizational structure, resources, and authority for making and implementing decisions. Most information about landscapes enters the system through the science component and most human effects on landscapes are mediated through the decision-making component, which sets rules and compels action. The dotted line represents unregulated interactions between humans and landscapes that are not part of the model but need to be acknowledged (e.g., off-road vehicle use).

The social model has several interacting components. First, there is a societal component involving the general public and stakeholders (i.e., industry, environmentalists, Indigenous people, and others). These voices articulate the values, priorities, and concerns that are at play (see Chapter 3). Social values and priorities are what ultimately enable and constrain conservation.

Second, there is a science component, including both biological and social sciences, that provides the facts, ideas, and predictions needed to make robust decisions (see Chapter 4). Science is also needed to identify threats to biodiversity and to devise solutions (in common with the biocentric model). Finally, the application of science ensures that conflicts among values are resolved in the context of what can realistically be achieved.

The third component is a decision-making system that is overseen mainly, though not exclusively, by the government. In the face of complex trade-offs, a formal, structured approach to decision making is vital for finding optimum solutions (Gregory et al. 2013). The main steps include clarifying the objectives, identifying management alternatives, predicting outcomes under the available alternatives, and selecting the best option (see Chapter 10).

The three core components of the social model exist within, and are supported by, an institutional framework that provides organizational structure and establishes lines of authority. The institutional framework identifies who does what and provides the resources needed to turn management options into management actions (see Chapters 6–8).

To be clear, the social model of conservation is not an alternative to the biocentric model; it is an extension of it. The core tasks of describing the natural state of biodiversity, and the actions needed to maintain it, remain central to the social model. We can think of these steps as defining the conservation need. The rest of the model places this need in a broader social and political context, which is central to implementation.

In practice, individual components of the social model are often missing or deficient (see the case studies in Chapter 11). Institutional structures may be fragmented, decision-making systems may be dysfunctional, scientific studies may focus on the wrong questions or be ignored, and so on.

CHAPTER II

THE HISTORICAL FOUNDATIONS OF CONSERVATION IN CANADA

The Historical Foundations of Conservation in Canada

A New World

The contemporary practice of conservation is rooted in the events, decisions, and learning that have occurred in the past. This applies not only to the landscape changes that now threaten many species, but also to our collective way of thinking about biodiversity and what it means to maintain it. To understand current conservation practice we need to understand its historical foundations. The aim of this chapter is to provide that foundation by tracing the evolution of conservation in Canada from the initial influx of Europeans through to the start of the new millennium. More recent developments will be discussed in subsequent chapters.

When the Europeans packed their bags for the New World, they brought with them a worldview that emphasized human dominion over the earth. European conservation practices were based on the control of land and resource use by nobility, and they were not part of a culturally shared worldview (Donihee 2000). Furthermore, in the battle for survival that characterized the lives of early settlers, wilderness was something hostile that needed to be subdued and tamed, not preserved. In any case, few could perceive the need for conservation in a land so bountiful and limitless.

The effects these early Canadians had on the environment grew with their numbers and with the expansion of the fur trade. Canada's population increased slowly at first, remaining under 50,000 until the mid-1700s. It reached 3.5 million by the time of Confederation in 1867 (SC 2014a). Three categories of activity accounted for most environmental impacts during this early period: hunting and trapping, agriculture, and tree harvesting.

The activity with the most widespread ecological impact was trapping associated with the fur trade. Beavers were the primary species of interest, and by the late 1800s, they had been extirpated from many parts of Canada. Given the beaver's role as an ecosystem engineer and keystone species, its removal had widespread ecological repercussions (Hood and Larson 2015). The Hudson's Bay Company eventually instituted trapping limits as a conservation measure; however, the directives were never effectively implemented (Sandlos 2013). What ultimately saved the beaver was not conservation but changing fashion. By the mid-1800s, beaver hats were out, and silk hats were in.

In contrast to the fur trade, which affected species and ecosystems across Canada, hunting, agriculture, and tree harvesting were concentrated near the early settlement areas. Before Confederation, almost all of these settlements were located along the St. Lawrence River, the Great Lakes Lowlands, and around the coasts of the Maritime provinces (Fig. 2.1). Agriculture had the greatest impact because it involved the clearing and transformation of land and because it supported an ever-increasing human population with an ever-growing environmental footprint.

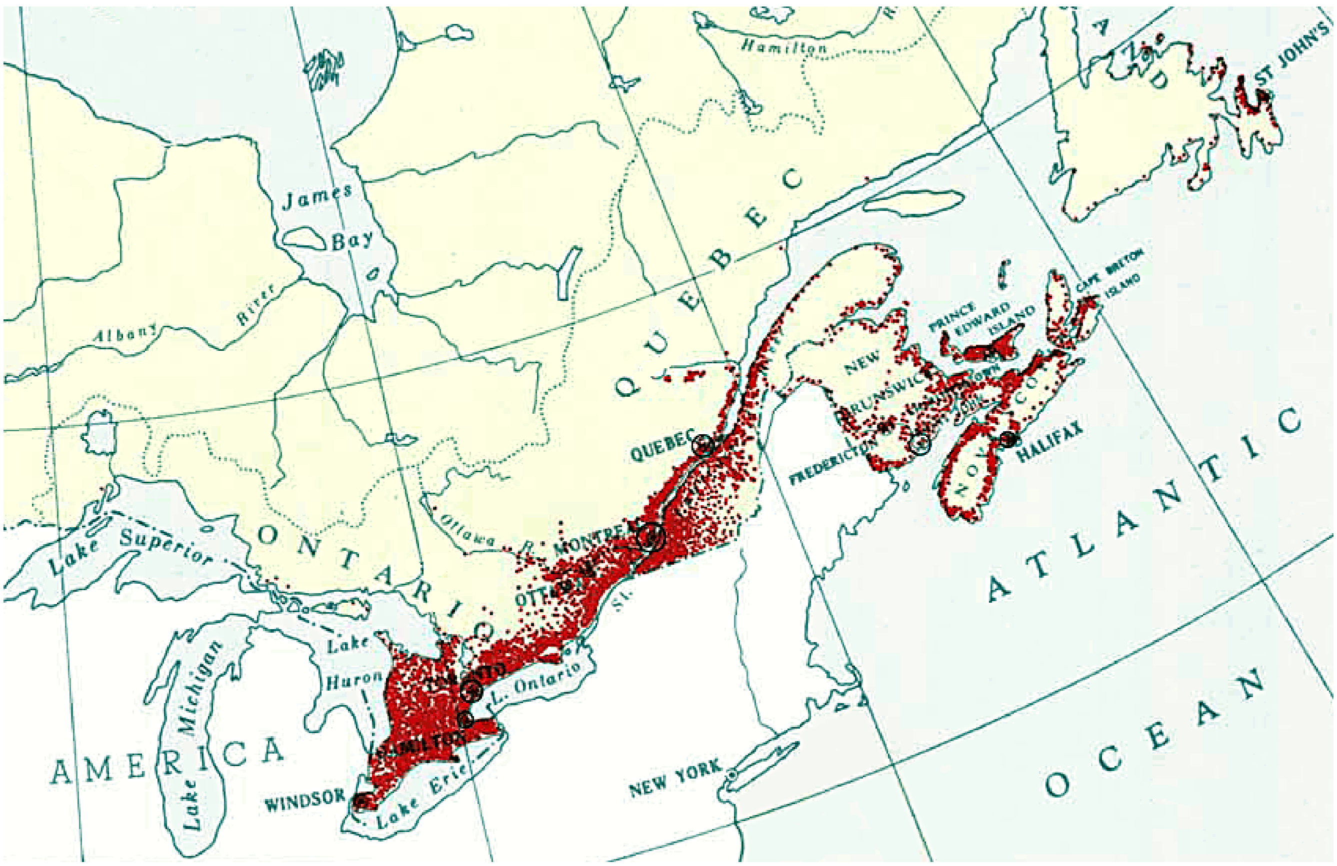


Fig. 2.1. The distribution of Canada's urban population in 1871. One dot represents 1,000 inhabitants. Source: Atlas of Canada, 3rd Edition (http://open.canada.ca).

Even though most settlers were not dependent on hunting for survival, supplemental hunting was common and resulted in substantial pressure on local wildlife. Hunted populations went into regional decline, and some species, like elk, were extirpated from some eastern areas (Rosatte 2014). Forests were also pushed back, as the need for agricultural land, lumber, and fuel for heating steadily increased. The eventual loss of 90% of southern Ontario's Carolinian forest, Canada's most diverse ecosystem, can be traced back to this period (Suffling et al. 2003).

Nation Building

With Confederation in 1867, Canada transitioned from a collection of British colonies to a country in its own right. From the perspective of conservation, the most important aspect of Confederation and the associated *Constitution Act* was the division of power between the provinces and the federal government. The provinces were awarded exclusive control of lands and resources within their boundaries and given primary responsibility for their management. Wildlife was not mentioned in the *Constitution Act* directly but has since been interpreted to be a component of the land, and therefore, is also under provincial control (Kennedy and Donihee 2006).

There are a number of specific provisions in the *Constitution Act* that create exceptions to the general rule of provincial control over wildlife. In particular, the federal government has control over the management of fisheries, most migratory birds, and endangered species. It also shares responsibility for various aspects of environmental management that impinge on conservation indirectly, such as the control of pollution and the environmental assessment of certain types of industrial projects. Finally, the federal government has retained partial control of land and resources in the territories and has full control over certain other areas, such as national parks.

Confederation was followed by a period of vigorous nation building. Settlement of the West was a top priority for the new national government and was supported by the building of a transcontinental railway and a campaign to draw immigrants from all corners of Europe with offers of free land. These efforts were highly successful in terms of their stated goals. By 1911, Canada's population had more than doubled from the time of Confederation, to 7.2 million (SC 2014a). Export markets grew in importance and began to include a wider range of products. Businesses were established, a service sector was developed, and urban centres grew in size and importance.

Economic growth and the great wave of immigration led to increased environmental degradation. The problems were similar to those of earlier periods but the rate of change was now much faster. In the space of only three decades (1881–1911), the area of farmland in the Prairie provinces increased from 1.2 to 24.3 million hectares, comprising over half of all farmland in Canada. Canadian wheat exports rose 16–fold, to 97.6 million bushels (SC 1983a). Not only were there more people in more places than ever before, but growing external markets for resources placed increasing and unsustainable demands on natural systems. Last but not least, the frontier mentality and human-centred worldviews of earlier periods remained largely intact, muting concerns over the ecological changes that were occurring.

An important feature of this period was the existence of markets for wildlife meat and parts, which increased the rate of harvest far above that needed to meet local subsistence needs. Unsurprisingly, targeted species declined precipitously. A prominent example is the plains bison, which once roamed the Great Plains in the millions. By the late 1880s, the Canadian bison population was extirpated and only a few hundred individuals remained in the US.

Market hunting of bison was initially conducted mainly by the Métis from Manitoba's Red River region (Dobak 1996). By the mid-1800s, their hunts had evolved into highly organized biannual events, sometimes involving over a thousand individuals. The bison meat provided winter provisions for Métis families and also supported a thriving trade with the Hudson's Bay Company and European colonists. The 1870s brought hide hunters, who "killed lavishly for the one or two dollars per mature hide that American tanners were prepared to pay" (MacEwan 1995, p. 59). In fairly short order, all that was left of the vast bison herds was their bones, which were later collected and ground up as fertilizer.

A similar fate befell the passenger pigeon (Fig. 2.2), which went from being the most abundant bird in North America to extinction in the late 1890s. These pigeons had always been hunted because their colonial nesting habits and large numbers made them an irresistible target. The tipping point to unsustainability occurred when hunting became commercialized and then increasingly mechanized in the late 1800s. By the end, railcars were annually shipping pigeons by the millions to markets in large cities (Yeoman 2014). Although market hunting was not the only factor involved in the demise of the passenger pigeon (Bucher 1992), extinction is unlikely to have occurred without it.

Fig. 2.2. A male passenger pigeon, displayed at Chicago's Field Museum of Natural History. Credit: J. St. John.

Trade in meat was not the sole focus of market hunting in the 1800s. The fur trade was still important at this time, and there was also a thriving whaling industry that was providing whale oil for lamps and baleen for corsets. This would eventually land many whale species on the endangered species list. Last, but not least—not to be outdone by the gentlemen and their beaver hats—ladies started a craze of their own involving the use of feathers to adorn their hats. Innocuous as this may seem, the growing size and affluence of human populations in the late 1880s generated an unsustainable demand, leading to the decline of many North American bird species, including Canada's own now-endangered piping plover (Doughty 1975). Wild bird feathers were also harvested for stuffing pillows, and this was one of the main drivers of the extinction of the Great Auk off the coast of Newfoundland in the mid-1800s.

Early Twentieth-Century Conservationists

Toward the end of the 1800s, the demise of the bison and passenger pigeon, and the overexploitation of many other species and other natural resources, began to affect the collective conscience of North Americans. Sporadic conservation efforts and localized restrictions on hunting had been implemented earlier, but these were of limited scope and were never effectively implemented (Loo 2006). What transpired at the turn of the twentieth century was a broad social movement that embodied a new way of thinking about wildlife and nature.

The first conservationists were mainly Americans. The end of the frontier was reached earlier in the US than in Canada, and environmental losses were more apparent, making the myth of limitless resources untenable (Foster 1978). Almost from the start, two disparate views of conservation emerged, and they remain distinct themes today: a **utilitarian** or "wise use" view and a **preservationist** view (MacDowell 2012). Advocates of the utilitarian approach, such as the first Chief of the US Forest Service, Gifford Pinchot, focused on the sustainability of resource use and elimination of wasteful practices. They also emphasized the importance of scientific management and centralized control over resource use.

The preservationists valued nature for its intrinsic qualities, rather than as a resource for human use. They were led by men such as John Muir, who co-founded the Sierra Club in 1892. The preservationists' main concern was the loss of wilderness, and their preferred tool was protected areas, where resource development was prohibited. Pinchot and Muir were both advisors to President Theodore Roosevelt, who was himself a strong advocate of conservation. Both views of conservation were advanced under his watch, though the utilitarian view was dominant and eventually co-opted the term "conservation."

Conservationist ideas percolating in from the US helped to generate a conservation movement in Canada, distinguished by strong support among political and business leaders (Sandlos 2013). The high-water mark was the establishment of the Commission of Conservation, through an Act of Parliament in 1909. The Commission was heavily influenced by Pinchot and his utilitarian views of conservation as well as ideas from the contemporary Progressive Movement about efficiency, science-based decision making, and professional management (Sandlos 2013). It published about 200 reports during its tenure, greatly expanding knowledge related to resource management and contributing to the development of public policy (MacEachern 2003). In so doing, it raised the profile and credibility of conservation and promoted its widespread adoption.

By World War I, Canada's approach to resource management had been completely overhauled. The state was now firmly in control, and the fragmented and uncoordinated management efforts of earlier periods had been replaced with top-down bureaucratic management systems involving planners, scientists, foresters, game wardens, and others. The new approach incorporated the concepts of utilitarian conservation and featured a legal foundation, professional staff, research-based problem solving, and effective enforcement. Attention was focused on three main areas: game management, forest management, and parks.

Game Management

The decline in wildlife populations during the 1800s was, fundamentally, a manifestation of the Tragedy of the Commons (Box 2.1). Human populations were now far too high and technology was far too lethal to maintain a sustainable rate of harvest in the absence of effective control mechanisms. This control was achieved by the early conservation movement, but not simply through tougher laws and regulations. The critical change was the emergence of a sport hunting ethic, originating mainly in middle and upper-class society (Loo 2006). In the absence of such a shared vision and ethic, it is unlikely that regulation alone would have been effective, given the challenge of enforcing such rules in Canada's vast wilderness.

Box 2.1. The Tragedy of the Commons

The Tragedy of the Commons is a resource management problem in which the users of a shared resource end up depleting it through the narrow pursuit of self-interest (Hardin 1968). In the absence of controls or assigned rights, individuals are motivated to take as much from the commons as they can because failing to do so means someone else may get their share. Perhaps the most grievous example in today's world is the global decimation of fish stocks through overfishing of the high seas.

By the turn of the twentieth century, Canadian society was changing, as cities grew and became the focus of political power. Subsistence hunting had no relevance for these urbanites, though many retained a strong desire to hunt and reconnect with nature as a recreational pursuit (Fig. 2.3). Sport hunting reached its pinnacle during this time and was one of the top recreational activities for men (Herman 2003).

Fig. 2.3. Portrait of a sport hunter, circa 1900. Credit: B. Hoare, Provincial Archives of Alberta.

The objectives of sport hunting are far removed from those of subsistence hunting. It is not the meat, but the hunting experience that is of highest priority. And this changes everything. Instead of focusing on the most effective and efficient means of killing, sport hunting is based around ideas like challenge and fair chase (Posewitz 1994). As a result, wildlife is most valuable while it is alive, not dead. Finally, from the perspective of sport hunters, subsistence hunters, market hunters, and hunters that did not adhere to the sport hunting creed were all unwanted competitors.

The sport hunters, being largely urban based, were politically well connected. In fact, politicians were as likely as not to be sport hunters themselves. Therefore, the system of game management that developed during this period was designed to serve the needs of sport hunters over other users. The new system of management was based on three core policies, which remain in place today: (1) the absence of a market for the meat and products of game animals; (2) the allocation of hunting rights by law, not birthright, social position, or land ownership; and (3) a prohibition on the frivolous killing of wildlife (Geist 1988). Earlier piecemeal hunting laws and regulations were also coordinated and strengthened, and game wardens were hired to ensure compliance. Practices contrary to the sport hunting ethic of fair chase were generally banned, and restrictions were placed on the number and types of animals that could be taken and on the timing of the hunt (Donihee 2000).

These policies and regulations had several effects. First, they removed value from dead animals and increased the value of living animals. They also ensured that the killing of wildlife was not economically rewarding, once the costs of equipment and travel were accounted for. In addition, the take of individual sport hunters was reduced to a sustainable level. Finally, the system made each citizen a shareholder in wildlife, with a stake in maintaining healthy populations. An important caveat was that management interest was squarely focused on game species above all others. Species that were perceived to be a nuisance, such as wolves and raptors, were still killed indiscriminately.

The new system of game management was very successful in terms of its stated objectives. After decades of widespread decline, the populations of most game species stabilized and began to recover (Geist 1988). In turn, hunting opportunities increased, and so did economic benefits and jobs associated with wildlife (e.g., outfitters and equipment suppliers). Many conservation organizations also came into being, providing political and material support for conservation efforts.

This is not to say that the new system was free from detractors. Rural people, in particular, chafed at the new restrictions imposed upon them by what they perceived as urban elitists (MacDowell 2012). Market hunters were, of course, none too pleased either, though declining wildlife populations had already reduced their prospects for profit. In any case, neither of these groups had the political power needed to stem the tide of change.

The new system of wildlife management, which emphasized public access to the resource and the absence of markets, was applied to most game species. However, furbearers and certain fish species were handled differently. For furbearers, sustainable commercial harvest was achieved, and continues to be achieved, by regulating access through exclusive-use traplines. This privatization of the resource kept interlopers out and encouraged trapline owners to harvest at a sustainable rate. In addition, the high rate of reproduction of furbearers, relatively low economic potential of trapping, and the labour-intensive and arduous nature of trapping, all contributed to keeping supply and demand in balance.

Commercial harvest was also maintained for a variety of fish species, but here the outcome was generally very poor in terms of sustainability. In large part, this was because the resource could not be effectively privatized—neither fish nor boats could be tied to defined locations. Thus, the Tragedy of the Commons manifested, exacerbated by progressive improvements in the efficiency of commercial fishing. We will review an example involving walleye fisheries in Case Study 5 (p. 293).

Forest Management

Forest harvesting underwent a rapid expansion during the 1800s, supported by a thriving export market to the US and England, as well as growing internal demand. The general approach to harvesting was "cut and move on," which propelled cutting crews down ever-smaller tributaries of the waterways needed to transport the timber to market (MacDowell 2012). The advent of railroads in the mid-1800s greatly improved access to backcountry forests, leading to further increases in the rate and spatial extent of cutting.

Forest harvesting was only loosely regulated throughout most of the 1800s. The main concern of governments was the extraction of rents and the control of competition through regulated access (Ross 1997). In contrast to the US, access to forests was generally provided through temporary leases rather than land sales, and this turned out to be a pivotal decision. Over the years, the retention of public land ownership in Canada has been a critical factor in advancing forest conservation.

The conservationists of the early twentieth century were not concerned about the commodification of forest products, as they were with wildlife. Their major worry was that forest depletion would lead to timber shortages, jeopardizing future economic development (Drushka 2003). This was conservation with a very strong utilitarian and economic orientation. Three main problems were identified that required attention: farmers, fire, and poor harvesting practices.

The primary tool for dealing with agricultural clearing was the establishment of forest reserves, where land clearing and human settlement were prohibited (MacDowell 2012). The basic idea was to allocate landscapes according to the uses for which they were most suitable. In some areas, the forest reserves were intended to also support watershed conservation.

Concerns about fire losses led to regulations on the use of fire and the deployment of fire rangers in many parts of the country. Rangers sought to prevent fires, especially from careless brush burning and sparks from trains. They were also expected to find and fight fires, to the extent this was possible at the time (Drushka 2003).

As for harvesting practices, the conservationists engineered a major overhaul, which included new measures to ensure forest regeneration, sustainable rates of harvest, and the prevention of waste (Ross 1997). In addition, under the influence of Pinchot and the Progressive Movement, management was thoroughly modernized. Formal bureaucracies dedicated to forest management were developed at the provincial and federal level, and professional foresters came into existence. Research into sustainable and efficient forest harvesting also got underway, led by the federal government's new Dominion Forestry Branch (1899), Canada's first Faculty of Forestry, at the University of Toronto (1907), and the Commission of Conservation (1909).

Parks

Another manifestation of the early conservation movement was the establishment of parks. Unlike the US, where wilderness preservation was an important driver of park establishment, Canada's first parks were created mainly for their utilitarian benefits. A good example is Ontario's Algonquin Park, established as the first provincial park in Canada in 1883. This park was created with three specific uses in mind (MacEachern 2003). Sport hunters sought a wildlife sanctuary to provide hunting opportunities. Logging interests sought a forest reserve where a secure supply of pine could be obtained. And municipalities sought the protection of the headwaters of several major rivers. Wilderness preservation and the conservation of biodiversity were notably absent as motivating factors.

The creation of Banff in 1885, Canada's first national park, also illustrates the mindset of the time. In this case, the primary interest was the commercial potential of tourism (MacDowell 2012). The government and the directors of the Canadian Pacific Railway recognized that the region's spectacular mountain scenery and the newly discovered hot springs would draw travellers from around the world. It took just three years for the 250-room Banff Springs Hotel to be built, and other mountain parks and Canadian Pacific Railway hotels soon followed. Although tourism was the main focus, additional revenue was sought from hunting, mining, and logging, all of which were permitted in the mountain parks in their early years.

In 1911, Canada formally established a Parks Branch, responsible for overseeing the expansion of the national parks system (Tanner 1997). The agency was led by James Harkin, who would become one of Canada's leading voices on conservation and the preservation of Canada's special places. Additional provincial parks were established during this period as well. Management efforts were primarily focused on creating the infrastructure needed to support tourism and recreation within the new parks. Additional efforts were directed at increasing wildlife populations and the prevention and control of fire (MacDowell 2012).

In addition to the new recreational parks, several wildlife reserves were established to support the rehabilitation of species that had been decimated through overharvest. The largest of these, at 44,800 km^2, was Wood Buffalo National Park, established in 1922. Some of the other reserves that were established around this time, such as the National Antelope Parks in Alberta and Saskatchewan, were later decommissioned after the target species had recovered (Foster 1978).

Rise of the Machines

The threats facing biodiversity underwent a fundamental shift in the twentieth century. Whereas wildlife declines in the nineteenth century generally involved someone setting a trap or firing a gun, the declines of the twentieth century were mainly the result of widespread habitat degradation from agricultural expansion and industrial development. The rapid growth and intensification of the resource sector was the result of several interacting factors:

- **Mechanization.** The farmers and lumberjacks of earlier periods had only muscle power and hand tools with which to push back the frontier, limiting the rate of change. The story of the twentieth century is one of increasing mechanization—more machines doing more things, with more efficiency, and more power.
- **Energy.** The increase in mechanization and expansion of the development frontier was supported by and dependent on an ever-increasing supply of easily transportable energy, primarily in the form of diesel and gasoline.
- **Access.** A defining feature of the twentieth century was the development of an extensive national transportation network that not only linked together Canada's far-flung settlements, but also provided the access needed to bring resources from remote areas to market (Fig. 2.4).
- **Innovation.** Technological advancement was rapid in the twentieth century, leading to greater effectiveness in finding and exploiting resources as well as increased profitability.
- **Population size.** Canada's population increased steadily over the twentieth century, increasing the demand for resources and providing the labour needed to extract them.
- **Export market.** During the twentieth century, Canada became one of the world's leading exporters of resource staples, especially to the rapidly growing US market. Market demand was, in turn, a strong driver of development and technological innovation.

Resource development also led to the establishment of mining and mill towns in remote areas that owed their prosperity and survival to the extraction of a single resource. In time, these towns would become a politically powerful constituency that supported industrial development.

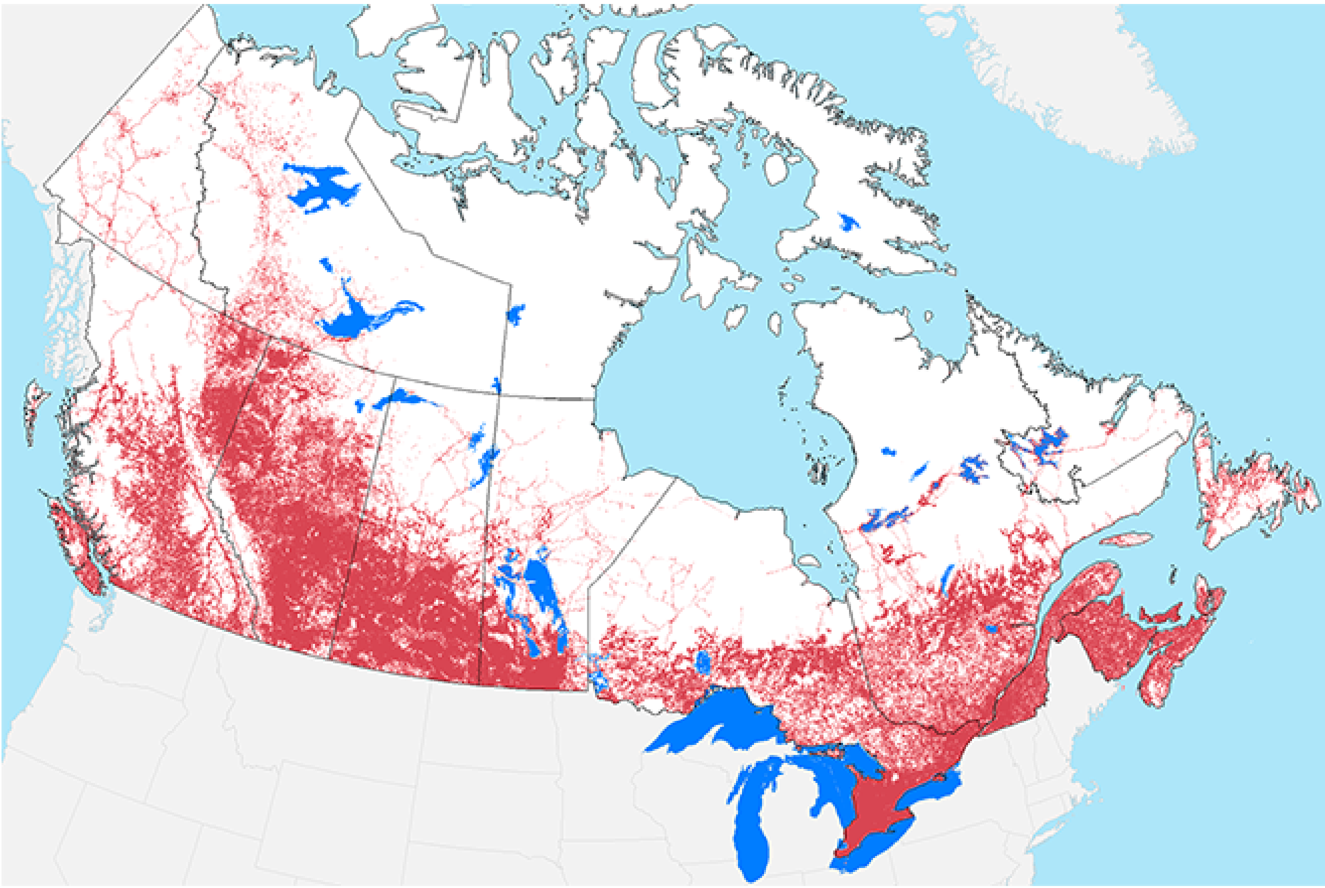


Fig. 2.4. The distribution of human access in 2013, based on Landsat imagery. Source: Global Forest Watch Canada.

Agriculture

The amount of land used for agricultural production reached its peak in the 1930s (Figs. 2.5 and 2.6). Most of the agricultural expansion occurred in western Canada, and the three Prairie provinces today account for over 80% of Canada's agricultural land. In the East, agriculture remained focused in the Great Lakes Lowlands and the lands adjacent to the St. Lawrence River.

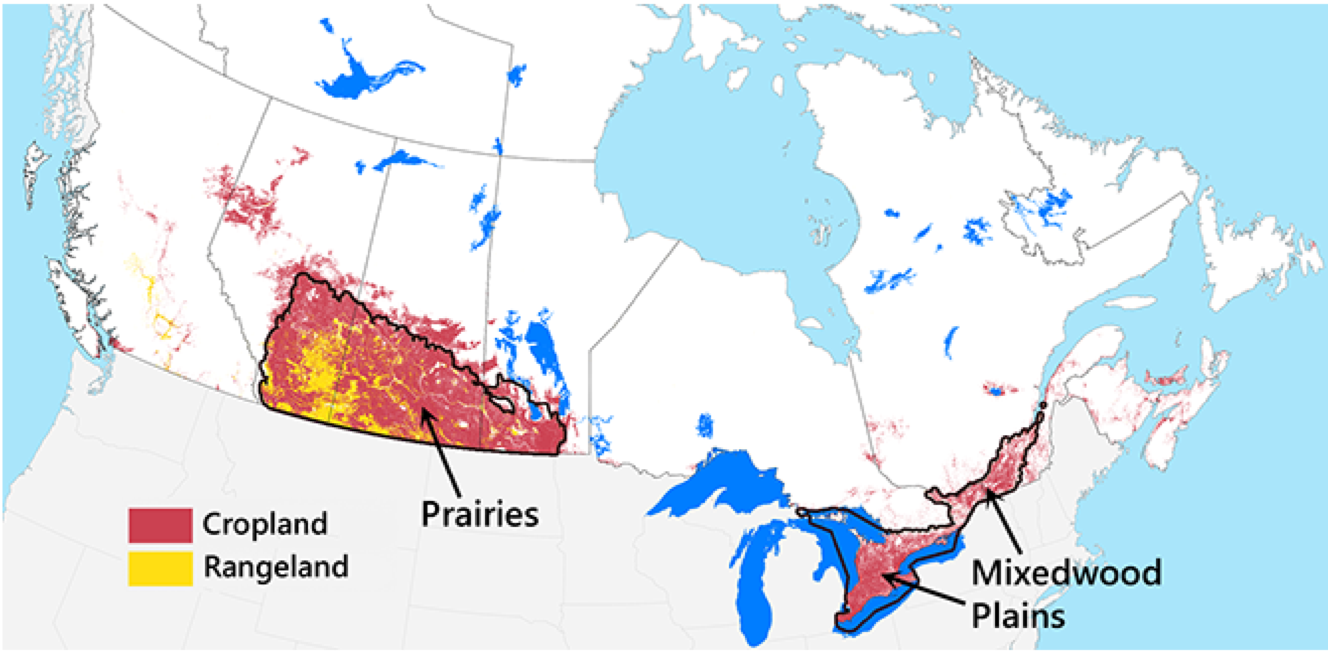


Fig. 2.5. The distribution of agricultural land in Canada in 2010. The Prairies Ecozone and Mixedwood Plains Ecozone are outlined in black. Source: Agriculture and Agri-Food Canada.

Although the amount of land devoted to agriculture plateaued early in the twentieth century, the impacts of agriculture on biodiversity continued to rise in later decades because of intensification. The transition from horses to tractors was pivotal. Steam tractors were already available at the turn of the century but widespread ownership of tractors did not occur until affordable gas-powered models became available in the 1940s (Fig. 2.7). Over the years, farms increased in size, through consolidation, and tractors grew larger to match. Whereas the popular 1937 Allis Chalmers Model B produced less than 20 horsepower, John Deere now sells eight-wheel behemoths that produce 620 horsepower and weigh more than 20 tonnes.

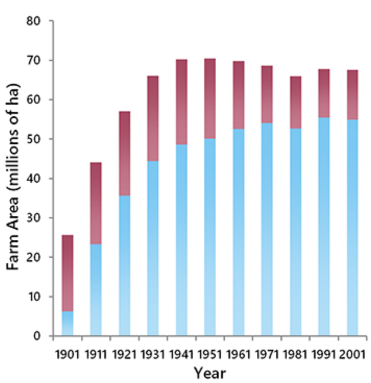


Fig. 2.6. The total area of farms in the Prairie provinces (blue) and the rest of Canada (red), 1901–2001. Source: Statistics Canada.

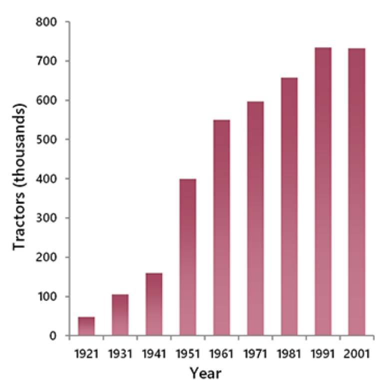


Fig. 2.7. The number of tractors on Canadian farms, 1921–2001. Source: Statistics Canada.

As farms mechanized, the cost of farming increased, leading to further intensification. Less and less of the landscape remained in a natural state. In addition, wetlands were drained to provide more cropland and to reduce the nuisance they represented to large farm machinery. It is estimated that more than 40% of prairie wetlands were lost to drainage over the past century and there is little evidence to suggest that the rate of loss has slowed in recent years (Cortus et al. 2011). Other manifestations of intensification included the removal of hedgerows, especially in Eastern Canada, and a progressive increase in the use of fertilizers, herbicides, and pesticides.

The rangelands of southeast Alberta and southwest Saskatchewan (Fig. 2.5) merit special mention because they followed a different trajectory. Low moisture inputs in these areas made them unsuitable for growing crops or pasture grass, so the native prairie remained largely intact. However, the replacement of bison with cattle, the control of prairie fires, and invasion by agronomic grasses had cascading effects on the integrity of these ecosystems (Fuhlendorf and Engle 2001). This region was also impacted by the development of an extensive road network.

Forestry

Rail networks expanded rapidly in the early decades of the twentieth century, providing access to a progressively larger proportion of Canada's merchantable forest. Another important change in this period was the rise of the pulp and paper industry, particularly in Eastern Canada (Kerr et al. 1990). Advancements in the design of the rotary press allowed the production of large numbers of daily papers, which became popular throughout North America. Demand for newspapers was also stoked by rising population levels.

Harvesting for pulpwood led to changes in forestry practices. Previously, trees were individually selected based on their suitability for producing dimensional lumber, which meant that much of a forest stand remained after harvesting. With pulpwood harvest, smaller trees could be utilized and so could species not suited for lumber production. Therefore, harvesting became more intensive and involved a greater range of stand types (Drushka 2003).

The next major change in forestry was mechanization, which became widespread after World War II. Most important was the internal combustion engine, which powered everything from chainsaws to large logging trucks. Trucks offered much greater mobility and flexibility than trains, and consequently, the development frontier was pushed even deeper into Canada's hinterland.

Mechanization also led to a further intensification of forest harvesting, culminating in the clearcut approach, which became the dominant method of harvesting in the last half of the twentieth century. Clearcutting offered several advantages for timber companies (Nyland 1996). First, it was efficient to implement, especially once harvesting was done with large machinery instead of chainsaws. Clearcutting also provided companies greater control over regeneration trajectories. Monocultures of desired species could be generated, boosting timber yields relative to natural regeneration, especially in mixedwood systems. The regenerating clearcuts were also even-aged, which facilitated the achievement of an even flow of timber each year at a standardized age at harvest (i.e., the rotation age).

The combination of mechanization and improved access led to steadily rising production of both lumber and pulp over the twentieth century (Fig. 2.8). The majority of Canada's merchantable forest is now subject to harvesting, and the remaining unallocated forest is mostly in the far north where productivity is low (Fig. 2.9). Harvesting has resulted in a progressive simplification of forest structures and patterns over time, and these changes continue to accumulate (see Chapter 5). Furthermore, forestry access roads have fragmented forested landscapes and served as wicks, drawing in other industrial and recreational users and their associated ecological impacts (Trombulak and Frissell 2000).

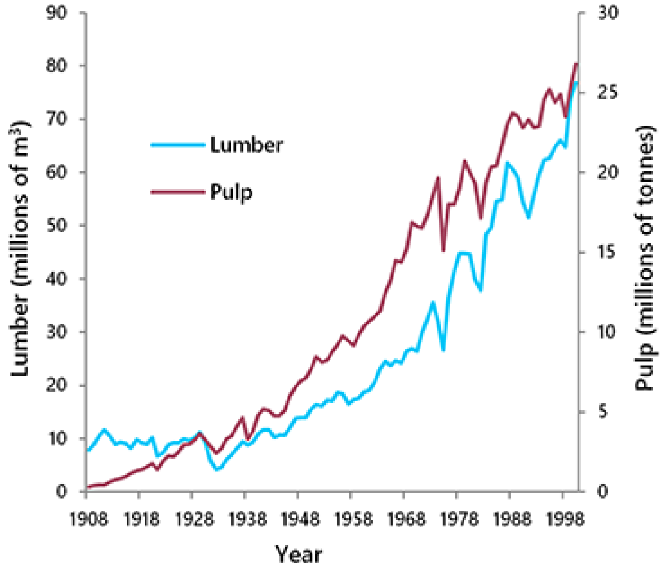


Fig. 2.8. The production of sawn lumber and pulpwood in Canada from 1908–2000. Source: Statistics Canada.

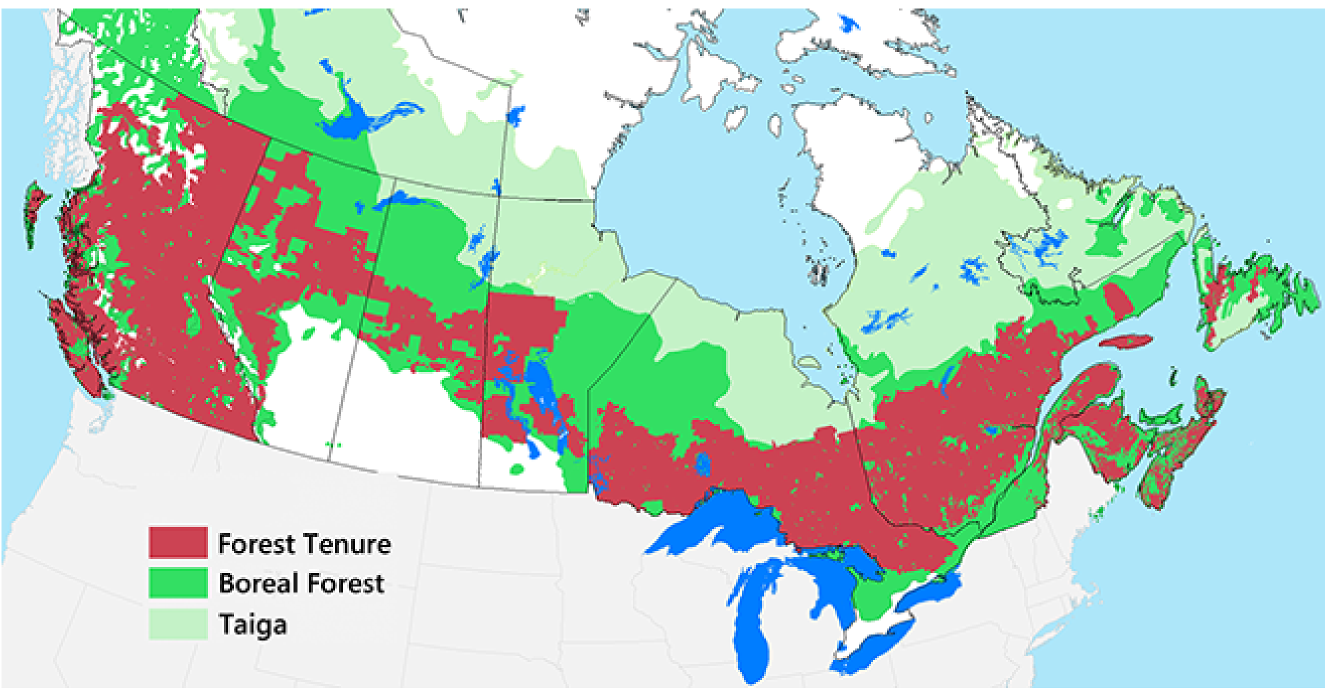


Fig. 2.9. The distribution of forest tenure (red) in 2013. The extent of forested land is shown in Green. Source: Global Forest Watch Canada and Canada's Forest Zone classification.

Oil and Gas

Oil and natural gas (Fig. 2.10) were discovered in Canada in the nineteenth century. However, significant levels of production did not occur until the middle of the twentieth century, when several factors came into alignment.

The first factor was demand, which increased exponentially once the internal combustion engine came into widespread use. The second was exploration success, which improved as a result of systematic seismic surveys and better understanding of subsurface geology. The third was the ability to extract and ship the oil and gas to market, which improved with better technology and expanding infrastructure.

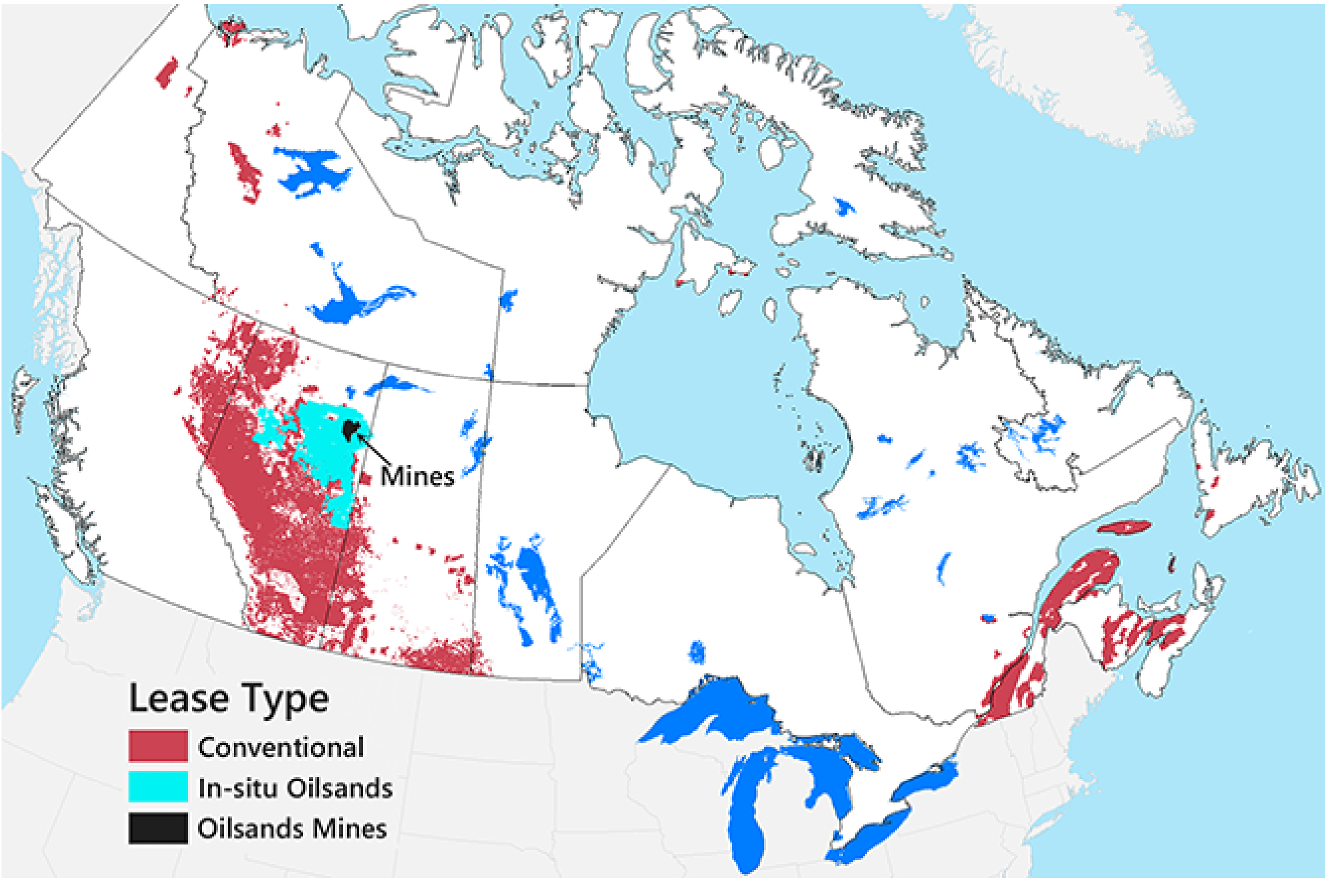


Fig. 2.10. The distribution of oil and gas tenure, by type, in 2013. Source: Global Forest Watch Canada.

Oil and gas deposits were eventually found throughout the Western Canadian Sedimentary Basin, stretching from northeast BC, across most of Alberta, and into southern Saskatchewan. Additional deposits were found in the Maritimes, Northwest Territories, and offshore. Drilling and infrastructure development were initially centred in the Prairie region, where most of the early discoveries were made and where access was plentiful. However, by the 1960s, road and pipeline networks were being developed deep into the boreal forest and the foothills of the Rocky Mountains. In areas of active oil and gas development, the annual rate of forest clearing for roads, well sites, pipelines, and seismic exploration approached the rate of cutting by the forestry sector (ECA 1979).

In 1973, the production of conventional oil reached its peak and then began to slowly decline (Figs. 2.11–2.12). Thereafter, growth in oil production was achieved through the development of unconventional deposits. Most important were the oil sands in northern Alberta, which contained thick bitumen mixed with sand. Some of the oil sands deposits were close enough to the surface to be recovered through surface mining, and this is where initial production began, in 1967. In the 1990s, technology was developed that allowed the recovery of deeper oil sands deposits using steam heating and in situ extraction. As a result, the land area affected by oil sands development expanded from 4,800 km^2 (surface mining only) to 142,000 km^2 (Fig. 2.10). Today, the oil sands produce more oil than all other sources in Canada combined (Fig. 2.12).

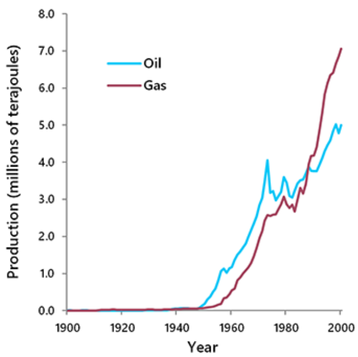


Fig. 2.11. The production of oil and natural gas in Canada from 1900–2000. Source: Statistics Canada.

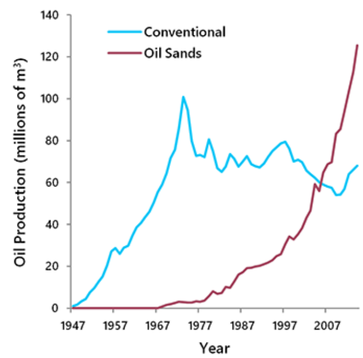


Fig. 2.12. The production of oil in western Canada, by type, from 1947–2014. Source: CAPP 2017.

Not all of Canada's oil and gas deposits have been brought into production. Many deposits remain stranded because of a lack of infrastructure and challenging working conditions, especially in the Northwest Territories. In areas with established infrastructure, there has been a tendency for successive waves of development to occur, as evolving technologies allowed different types of deposits to be profitably extracted.

The development of oil and gas over the last century has left a significant cumulative footprint, especially in the western boreal forest. Over 400,000 wells were completed in Canada between 1955 and 2017 (CAPP 2017), disturbing approximately 1 ha of habitat in each case. Virtually all of these wells required the construction of an access road, and most were connected to a pipeline. In addition, hundreds of thousands of kilometres of seismic cut-lines remain in forested areas as a legacy of exploration activities. Oil and gas development and refinement also resulted in air, soil, and water pollution (see Chapter 5).

Mining

Before 1900, mining in Canada was limited to small-scale operations focusing mainly on coal, iron, and gold (Cranstone 2002). Mining slowly expanded in the early twentieth century, closely tied to the expansion of transportation infrastructure. Many mining towns in the Canadian Shield and BC interior got their start during this early period, including Kimberley, Flin Flon, Sudbury, and Val-d'Or.

The demand for metals and other minerals rose rapidly after World War II as a result of increased mechanization. Demand was further stimulated by advances in metallurgy, which led to new applications for metals. Rising demand provided mining companies with the incentive and security needed to undertake large, capital-intensive mining projects. Moreover, with the advent of heavy machinery, it became possible to remove large quantities of surface material to access extensive low-grade deposits through open-pit mining. Finally, advancements in science and technology enabled systematic exploration for mineral deposits and provided better methods of ore refinement. Consequently, mining production increased rapidly in the second half of the twentieth century (Fig. 2.13).

Today, there are 220 principal mines in Canada producing more than 60 minerals and metals (NRCAN 2013). These mines are distributed across the entire country, including the territories (Fig. 2.14).

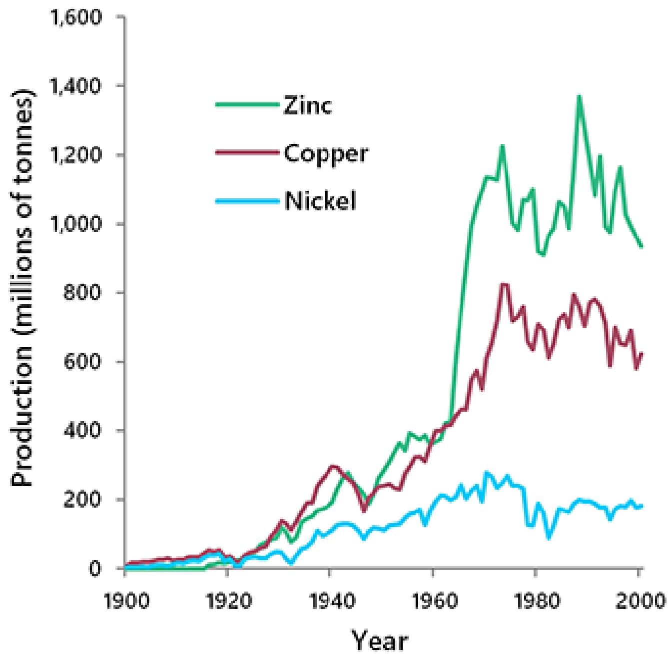


Fig. 2.13. The production of zinc, copper, and nickel in Canada from 1900–2000. Source: Statistics Canada.

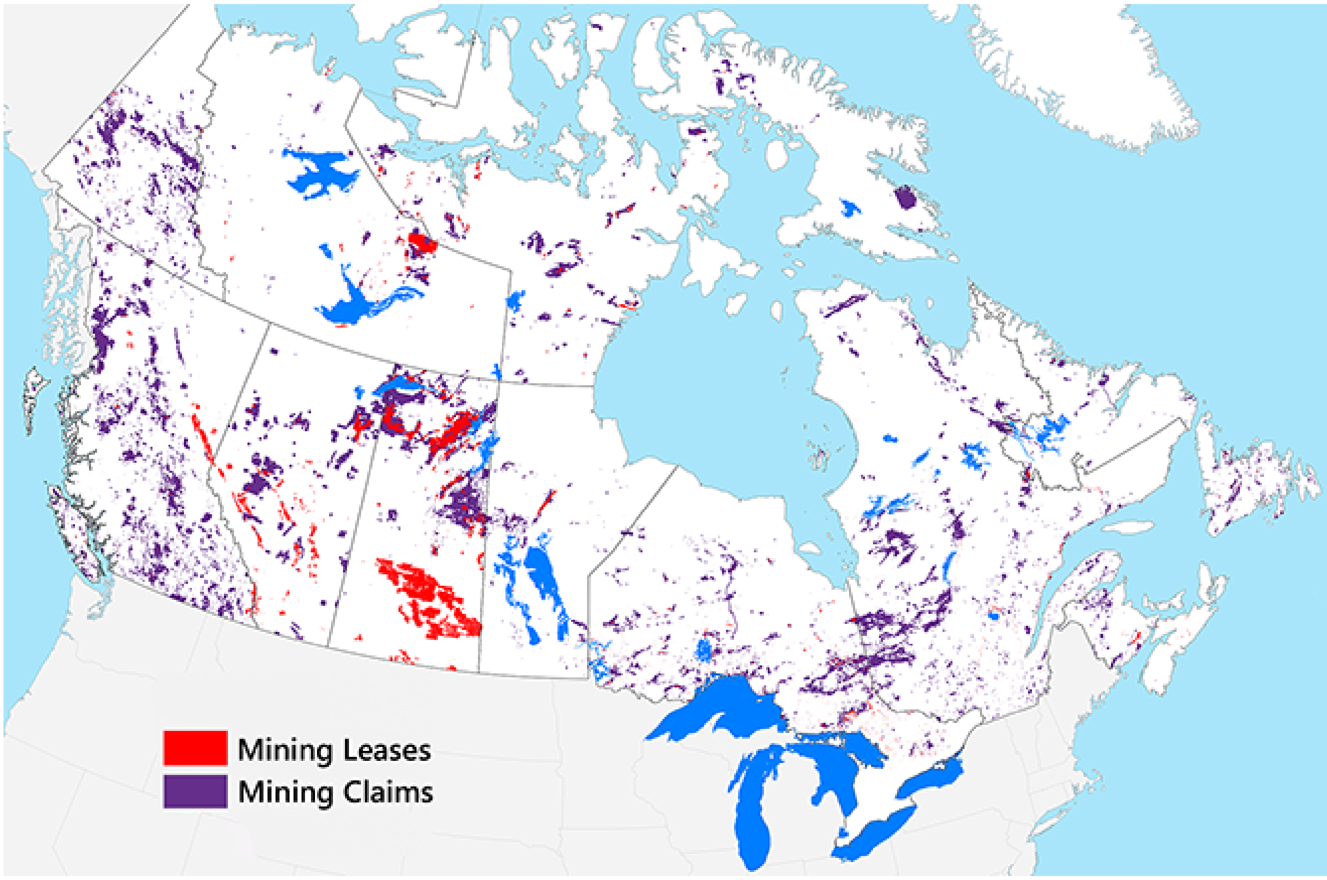


Fig. 2.14. The distribution of mining tenures, by type, in 2016. Source: Global Forest Watch Canada.

From an ecological perspective, the most important legacy of mining in the twentieth century is the waste produced. Hundreds of millions of tonnes of rock had to be crushed, ground, and then chemically processed to extract the target minerals, which were generally a minor component of the ore (less than 1% for many metals). The residual tailings were stored on-site or discharged into the water, posing a variety of environmental hazards (Allan 1997).

Ontario's Sudbury region provides one of the more egregious examples of environmental harm caused by mining in the twentieth century. Over 7,000 lakes within a 17,000 km^2 area were acidified to the point of significant biological damage (Keller et al. 2007). In addition, the lakes and soils in the Sudbury region accumulated dangerously high levels of copper, nickel, zinc, and lead from windblown dust from tailings piles (Nriagu et al. 1998). The result was "an unusual anthropogenic ecosystem of denuded barren land with lifeless lakes" (Nriagu et al., 1998, p. 99).

Not all mining operations were as bad as Sudbury's, but it was not alone in leaving a long-lasting environmental legacy. Over 10,000 abandoned mines exist in Canada (MacKasey 2000), many of which are leaching arsenic, mercury, lead, sulfuric acid and other chemicals into the environment (Parsons 2007). Consequently, twentieth-century mining continues to have an environmental impact today that extends well beyond the footprint of the mines themselves.

The Advent of Modern Conservation

The ecological deterioration and decline in wildlife populations that occurred as a result of industrialization in the twentieth century presents a question: what happened to the early conservationists? Some authors have suggested that the initial flourish of interest in conservation at the turn of the twentieth century waned once societal attention shifted to economic growth in the 1920s (MacEachern 2003; MacDowell 2012). In reality, conservation efforts during this period did not diminish at all, they expanded and became institutionalized (Burnett 2003). However, they remained narrowly focused on species that were hunted or harvested; broader conservation concerns had yet to be meaningfully recognized.

Over the ensuing decades, we got more of everything—bureaucrats, game wardens, foresters, scientists, schools, and associations—greatly expanding management capacity. Our knowledge base also improved. By the 1930s, game management emerged as a distinct discipline and research was well underway into species distributions, population sizes, food and habitat requirements, predator-prey dynamics, disease, and many other topics. Silviculture likewise underwent substantial development and maturation.

As capacity increased and ecological knowledge accumulated, management efforts became increasingly sophisticated. The basic objective of sustainable use morphed into the concept of maximum sustained yield, which guided research and management efforts in wildlife and forestry for much of the century (Larkin 1977). Regulations to avoid overexploitation were fine-tuned, and steps were taken to increase the productivity and long-term sustainability of desired resources. Species with no direct utility were largely ignored, and those identified as having negative effects were often targeted for elimination.

The management of Wood Buffalo National Park during the first half of the twentieth century provides a window into the mindset of the time. The park was established in 1922 to support the recovery of bison. Initial management efforts simply involved a prohibition on hunting by local Indigenous communities and others. Once the herd began to recover, the park administration began a program of small-scale, seasonal bison hunts, in response to a request for bison meat from the local residential school (McCormack 1992).

From the early 1940s until well into the 1950s, wolves in the park were poisoned with strychnine and cyanide, and a wolf bounty was used to encourage trapping, all to increase the production of bison (Fuller and Novakowski 1955). In the early 1950s, infrastructure within the park was expanded, and the commercial slaughter of bison began in earnest, lasting until 1967. In total, several thousand buffalo were killed, along with an unknown number of wolves (McCormack 1992). Wood Buffalo National Park forests fared no better. Approximately 70% of the park's riparian old-growth spruce was clearcut between 1951–1991, without concern for the species that depended on it (Timoney 1996).

The upshot is that modern concepts of biodiversity conservation did not arise through the progressive refinement of early twentieth-century conservation principles. Management capacity and knowledge certainly increased over the years, but the objectives of management remained wedded to a narrow, utilitarian view of conservation. For the most part, wildlife and forests were treated as commodities, even in parks, and progress was measured in terms of rising production.

To be fair, there were some individuals at the fringe who argued for a less utilitarian approach to resource management. They were unable to effect much change during their time, but they did help prepare the ground for the future. One of these individuals was Grey Owl, whose articles and books were popular in the 1930s (Loo 2006). Writing from a cabin in northern Saskatchewan, Grey Owl railed against the commodification of wildlife. He suggested that conservation was hampered by the view that nature was a basket of goods that could return an income if properly managed.

Another important figure was Aldo Leopold, considered to be the father of wildlife management. Leopold's early career involved killing cougars, wolves, and bears in New Mexico. However, in his later years, he came to believe that these types of management activities were misguided. In his most influential work, the *Sand County Almanac* (1949), he outlined a biocentric approach for interacting with nature that he termed the "land ethic." The non-consumptive values and holistic ecosystem-based management concepts he articulated presaged the future direction of conservation:

> The land is one organism. … If the biota, in the course of aeons, has built something we like but do not understand, then who but a fool would discard seemingly useless parts? To keep every cog and wheel is the first precaution of intelligent tinkering. … The land ethic simply enlarges the boundaries of the community to include soils, waters, plants, and animals, or collectively: the land. A land ethic of course cannot prevent the alteration, management and use of these resources, but it does affirm their right to continued existence. (pp. 190, 239–240)

A notable Canadian figure of this period was Ian McTaggart-Cowan. He advanced a holistic approach to conservation through television, radio, writing, and lectures. Other individuals and groups promoted direct interaction with wildlife. Birdwatchers and field naturalist groups were in the forefront (e.g., establishing the Audubon Society of Canada in 1948). Low-level efforts to support species at risk of extinction also continued. These efforts expanded from their initial focus on overhunted game species to new species such as the whooping crane and trumpeter swan. At the international level, the International Union for the Conservation of Nature (IUCN) was established in 1948, with a primary focus on endangered species.

Origins of the Environmental Movement

Although Leopold and his compatriots influenced many people, they were too far ahead of their time to affect mainstream thinking. The real crucible of modern conservation was the environmental movement of the 1960s, which carried conservation along like a surfer on the crest of a wave.

The environmental movement arose as a collective response to the negative impacts that industrialization was having on the environment. But there is more to the story than simple cause and effect. Consider, for example, the Cuyahoga River which runs through Cleveland, Ohio. This river was so polluted with industrial waste that in 1969 it started on fire (Stradling and Stradling 2008). The fire attracted widespread media attention, including an article in *Time* magazine which reached millions of readers. It graphically illustrated just how bad the nation's environmental problems had become and fuelled outrage and demands for action. It was followed, in 1972, by the US *Clean Water Act*.

The problem with this narrative, which suggests a direct relationship between environmental damage, public concern, and policy response, is that the Cuyahoga River had burned at least nine times before (Stradling and Stradling 2008). The 1969 fire was not even the worst. The picture used in the Time article was actually from a much more serious fire in 1952. If environmental degradation was the trigger for action, why then did it take until 1969 for the public to engage? The same disconnect exists for most of the other environmental issues that rose to prominence in the 1960s. Clearly, other factors were at play. And in the messy details lie the foundations of modern conservation.

The world did not suddenly fall apart in the 1960s. Instead, a tipping point was reached that led to a new way of looking at things. In short, we developed an environmental consciousness. The key players in the development of this new environmental awareness included researchers, the media, environmental groups, policymakers, the public … and the hippies.

Hippies are symbolic of the counter-culture revolution that took place in the 1960s. Their contribution was to question authority (Fig. 2.15). Such youthful rebellion was, of course, not new. But in this case, many of the issues being raised resonated with the broader public, including the deaths of young men in an unpopular war in Vietnam, the prospects of a nuclear Armageddon, and slow poisoning from environmental pollutants. Consequently, many began to reconsider the merits of the paternalistic system that controlled decision making.

Fig. 2.15. A "flower-power" protest against the Vietnam war, in 1967. Credit: A. Simpson.

The range of issues attracting attention quickly expanded and people from all walks of life became activists or supporters of change. It was a social awakening, and North American society was never the same afterward. In particular, elitist, closed-door decision making would no longer be accepted. Henceforth, the public would demand a say.

The development of environmental consciousness also involved a conceptual frame shift. **Frames** are mental constructs that shape the way we see the world (Lakoff 2004). They help us make sense of events and information by providing background context and default interpretations of cause and effect. They are also value-laden, which means that certain aspects of reality may be highlighted while others are marginalized or ignored (Reese 2001). Because they are mental constructs, frames can change over time, even if the underlying reality does not.

Prior to the 1960s, people did not think of the environment in the same way we do now. Most environmental deterioration occurred out of view, and relatively few individuals had any direct knowledge of what was happening. There were no government monitoring programs, no environmental reporters, no activist groups, and little scientific research on environmental problems. Incidents like the early Cuyahoga River fires were reported as isolated local events rather than symptoms of a broader problem. The existing frame, to the extent that one existed at all, was that environmental damage was the cost of progress (Sachsman 1996).

The initial change in perspective was led by individual scientists with a personal interest in the environment and by environmental activist groups, most of which were spawned by the broader counter-culture revolution. These individuals and groups gave voice to the environment, bringing firsthand accounts and analysis of what was happening to a public that was unable to witness the changes directly. The publication of *Silent Spring* in 1962, by Rachel Carson, was a seminal event. It drew attention to the effects that pesticides were having on birds and, more generally, to the powerful and often negative effects of humans on the natural world.

As the 1960s progressed, the media began to play a central role in facilitating the environmental dialog, linking information providers with the general public. Stories about the environment proliferated and journalists began to connect the dots, interpreting individual local events in the context of broader national-scale concerns. By the time the Cuyahoga River burned in 1969, it was a national story about industrial pollution out of control, not a minor article in the local paper about how much it would cost to repair the railway bridge.

The interactions between scientists, activist groups, the media, and the public were mutually reinforcing. Mass media sparked public interest, which produced more activists and stimulated more scientific research, resulting in more information for the media in a virtuous cycle. In addition, political figures began to understand that taking a stand against pollution and other forms of environmental degradation made for good public relations. Their pronouncements and actions helped to legitimize the issues. Environmental awareness rose quickly, and by the first Earth Day, in 1970, the transition to the modern framing of the environment was essentially complete.

Indigenous Influences

Fig. 2.16. Grey Owl feeding a beaver. Credit: Canadian National Railways; Library and Archives Canada.

Indigenous perspectives on conservation first attracted public attention with the writings of Grey Owl, who gained a wide audience in the 1930s (Fig. 2.16). Grey Owl suggested that much could be learned from the way that Indigenous people interacted with nature (Loo 2006). Other writers, such as Henry Thoreau and John Muir, had presented conservationist ideas ahead of him, but Grey Owl was the first high-profile author to make a strong connection between conservation and Indigenous ways of life.

Although Grey Owl planted a seed, the time was not yet ripe for widespread uptake of Indigenous perspectives. This had to wait until the arrival of the environmental movement in the 1960s. In the search for alternative approaches, Indigenous worldviews re-emerged and found fertile ground.

The incorporation of Indigenous perspectives in this period centred on broad philosophical themes about stewardship and respect for nature that resonated with an increasingly environmentally aware public. These ideas were widely circulated, sometimes ending up on posters alongside Indigenous people and art (Fig. 2.17). Common themes included respect and reverence for wildlife and nature, the idea that resources are being held in trust for future generations, and the intrinsic interconnectedness and sacredness of animals, humans, and the land. Popularized quotes from a speech given by Chief Seattle in 1854 provide an example of how these ideas were presented:

> We know that the white man does not understand our ways. One portion of the land is the same to him as the next, for he is a stranger who comes in the night and takes from the land whatever he needs. The earth is not his brother, but his enemy—and when he has conquered it, he moves on.
>
> Humankind has not woven the web of life. We are but one thread within it. Whatever we do to the web, we do to ourselves. All things are bound together. All things connect.
>
> Humans merely share the earth. We can only protect the land, not own it.

A defining feature of this period was that Indigenous worldviews were being interpreted and presented mainly by non-Indigenous commentators. Ironically, Indigenous people were themselves still marginalized at the time (e.g., the right to vote was not awarded until 1960). It was the harmony-with-nature ideal that Indigenous people represented, and the powerful symbolism they provided, that was most important to non-Indigenous conservationists. To some extent, this involved filtering, simplifying, and romanticizing Indigenous culture and worldviews.

Fig. 2.17. An example of the types of posters that became popular in the 1970s, depicting Indigenous images alongside quotes attributed to Chief Seattle. Photo credit: H. Pollard, Provincial Archives of Alberta.

There was also liberal use of artistic licence concerning attribution. For example, although Chief Seattle did give a speech in 1854, his popularized quotes were later attributed to a television scriptwriter named Ted Perry (Stekel 1995). And while Grey Owl did live with and learn from Indigenous people, he was later exposed as an Englishman. Despite the dubious morality of some of these tactics, their historical impact is clear. Indigenous perspectives on nature went from relative obscurity into the mainstream, affecting the thinking of Canadians at large and helping influence the changes in resource management that occurred during this period.

Later in the twentieth century, Indigenous communities found their own voice and began to engage directly in public discourse about resource use, especially at the local level (see Chapter 3). Moreover, conservation was often subsumed into broader discussions about treaty rights, self-determination, and control over resources.

An important feature of this later period was the development of place-based conservation campaigns involving formal alliances between Indigenous communities and conservation groups. Notable examples include campaigns against logging in Clayoquot Sound in British Columbia (Nuu-chah-nulth, 1993), Alberta's boreal forest (Lubicon Cree, 1990), and the Temagami forest in Ontario (Teme-Augama Anishnabai, 1989). The objectives of conservationists and Indigenous communities aligned in the context of these campaigns and this provided the foundation for widespread gains in wilderness protection.

From Game to Biodiversity

Although the primary concern of the early environmental movement was pollution, the plight of wildlife also received increased attention at this time (Fig. 2.18; Gregg and Posner 1990). The scope of concern expanded beyond overexploitation to include habitat degradation and the harm to wildlife arising from pesticides and pollutants. Public attitudes toward wildlife also changed, with increasing emphasis placed on non-consumptive values, the moral right of all species to exist, and general respect for nature.

Narrow utilitarian perspectives were faulted for failing to prevent the environmental declines that had occurred in preceding decades. From this point forward, resource management would be increasingly scrutinized and contentious, involving competing and conflicting values held by different segments of society. Utilitarian values continued to play an important role in decision making but they were no longer the default.

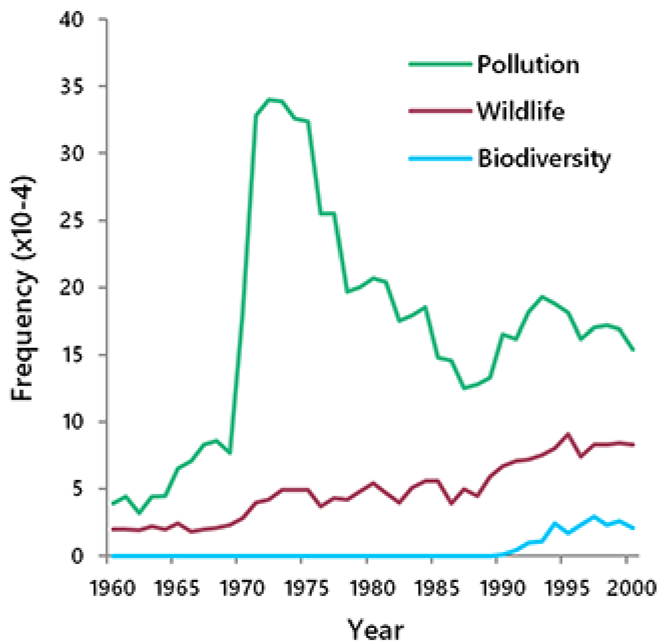


Fig. 2.18. The use of the words "pollution," "wildlife," and "biodiversity" in North American books, 1960–2000. Source: Google ngram viewer.

The changing status of the wolf is illustrative of the public's evolving attitudes toward wildlife. Prior to the 1960s, wolves were considered dangerous and undesirable, a menace to human safety and livelihood. Extermination campaigns, such as those in Wood Buffalo National Park, were routinely conducted, and public opposition was basically nonexistent. In the 1960s, through writers such as Farley Mowat and filmmakers like Bill Mason, the public—especially the urban public—began to see wolves in a new light. In Mowat's *Never Cry Wolf*, released in 1963, wolves were noble creatures whose commendable conduct highlighted the virtues of nature (Loo 2006). Mowat may have made liberal use of literary licence, but his story resonated with millions of readers. Mason was later hired by the Canadian Wildlife Service to provide some balance to Mowat's writing, but his 1972 documentary film, *Cry of the Wild*, further advanced the preservationist perspective.

Never Cry Wolf and *Cry of the Wild* were not just arguments against predator control; they embodied a new conception of wildlife and conservation (Loo 2006). Rather than efficient use, they advocated an ethic of existence, similar to what had been proposed earlier by Leopold. However, rather than emphasizing ecological integrity, their arguments were based on the intrinsic value and rights of animals. These views may have found little support among farmers and hunters, but had great appeal to city dwellers, who sided with the wolves. For these people, utilitarian and scientific arguments were not critical factors. They were swayed by the ethical dimensions of the issues, viewed in the broader context of social change and progressive loss of wilderness. For many, saving the wolf was a proxy for saving the wild.

The shift in public attitudes toward wildlife led to a series of policy changes. The US was again first to respond. However, this time Canada did not simply follow the US lead. Our response was substantially slower and differed in several important aspects that set us on a distinctly different policy trajectory (VanNijnatten 1999).

In the US, the landmark change was the passage of the US *Endangered Species Act*, in 1973. This Act was heavily influenced by input from scientists in the Bureau of Sport Fish and Wildlife and members of the conservation community (Illical and Harrison 2007). Through their efforts, the Act included a broad definition of species, a scientifically based determination of endangerment, and mandatory prohibitions on harm to listed species.

Notably lacking in the Act were economic considerations, reflecting the virtual absence of input or opposition from the private sector (Illical and Harrison 2007). In hindsight, many of the Act's provisions should have been red flags for the business and agricultural communities. However, lacking experience with such legislation, the private sector did not grasp the full import of the new Act as it related to their interests. In the absence of arguments to the contrary, the bill received near unanimous consent in the House and Senate.

A key feature of the US *Endangered Species Act* is the use of non-discretionary language, which reflects the separation of powers within the US system of government. Congress tends to be distrustful of the Executive Branch, which it must rely on to execute its instructions. Therefore, US environmental statutes have invariably employed non-discretionary language and firm deadlines to control the actions of administrative agencies, backed for good measure by "citizen suit" provisions that invite any individual to sue the executive should it fail to fulfill Congressional mandates (VanNijnatten 1999). The US system also contains many veto points which make it difficult to unwind laws once they are passed.

Once the practical implications of the *Endangered Species Act* began to be understood, developers sought to avoid them. This led to legal action, culminating in a Supreme Court challenge over the construction of a dam that posed a threat to a small endangered fish—the snail darter. The Supreme Court ruled that, despite the obscurity of the snail darter, the intent of the law was quite clear and non-discretionary: all species were to be protected, regardless of the cost. Amendments to the law were made in subsequent years, providing exceptions; however, there has never been enough support for the fundamental features of the Act to be repealed (Illical and Harrison 2007).

The trajectory of wildlife policy in Canada has been quite different from that of the US, for a variety of reasons (VanNijnatten 1999; Illical and Harrison 2007). In Canada, the legislative and executive branches of government are combined, so there is no incentive for creating non-discretionary laws. Our environmental statutes typically authorize, but do not compel, government actions. Second, because of decisions made at the time of Confederation, provinces have primary jurisdiction over natural resources, including wildlife. This has led to the uneven development of wildlife policy across the country and has hindered the coordination of conservation efforts. Finally, because Canada did not react as quickly as the US to the initial wave of environmentalism, there was an opportunity to learn from the US experience. The most important lessons were gleaned by the business community who, in contrast to their American counterparts, mounted a strong lobby to limit the scope and economic impact of Canadian wildlife legislation as it was being developed.

Initial efforts to update Canadian wildlife policies began in the mid-1960s, with efforts by the Canadian Wildlife Service, in cooperation with the provinces, to develop a national policy on wildlife. These efforts culminated in the passage of the *Canada Wildlife Act* in 1973—the same year as the US *Endangered Species Act*. The new Act expanded the definition of wildlife to include any non-domestic animal and also stated that any provisions respecting wildlife extended to wildlife habitat. The Act also included a provision for the protection of species at risk of extinction, expanding the scope of federal interest in wildlife beyond its traditional bounds. In contrast to the US law, there was no explanation of what the species recovery measures might entail, who would do them, or when they would be implemented. Instead, our Act simply stated, "The Minister may … take such measures as the Minister deems necessary for the protection of any species of wildlife in danger of extinction" (GOC 2015, Sec. 8).

Although a number of conservation groups and some members of Parliament were pressing for federal endangered species legislation, it was evident to Canadian Wildlife Service officials that such an approach would be anathema to the provinces (Burnett 2003). Therefore, national efforts were instead focused on a program to determine species status, without infringing on the legal prerogative of each province to manage wildlife within its boundaries. This led to the establishment of the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) in 1977.

During the 1980s, wildlife policy continued to evolve through regular conferences of federal and provincial wildlife ministers. One notable change was a further broadening of the definition of wildlife to include all wild organisms, including plants and invertebrates. In 1988, Canada's wildlife ministers established the Recovery of Nationally Endangered Wildlife committee to coordinate the development and implementation of recovery plans for the growing list of species that were being listed by COSEWIC. The committee was also intended to prevent species from becoming threatened or endangered and to raise public awareness of species conservation.

In the late 1980s, Canadians went "green," amid a renewed surge in global environmentalism. Polling in 1990 found that 82% of Canadians agreed with the statement "We must protect the environment even if it means increased government spending and higher taxes" (Lance et al. 2005). Also, Canadian Wildlife Service surveys demonstrated that wildlife-related activities, especially non-consumptive ones such as photography and bird-watching, were growing rapidly (Burnett 2003). Federal interest in conservation reached its high-water mark at this time. In 1990, the federal Progressive Conservatives unveiled their *Green Plan*, which provided funding for a range of environmental initiatives, including wildlife conservation. Canada was also an active participant in the development of the 1992 UN *Convention on Biological Diversity*, and we became the first industrialized country to ratify it. This was followed, in 1995, by the *Canadian Biodiversity Strategy* (EC 1995).

The *Canadian Biodiversity Strategy* marked the final stage in the conceptual evolution of conservation in Canada. In contrast to previous conservation policies, wildlife was now mentioned only in passing. The primary focus had shifted to biodiversity, a term that had only come into widespread use a few years earlier (Fig. 2.19). The Strategy defined biodiversity as "the variety of species and ecosystems on earth and the ecological processes of which they are a part" (EC 1995, p. 5). This was an important conceptual shift. Conservation was now about maintaining biodiversity, not the wise use of a few preferred species.

In the mid-1990s, efforts also finally got underway to develop federal species at risk legislation. In contrast to the 1973 US *Endangered Species Act*, which passed swiftly with minimal opposition, the development of Canada's *Species at Risk Act* (SARA) was highly contentious. Conservation groups were guided by the US experience and sought comparable mandatory provisions for endangered species in Canada. However, business interests, also guided by the US experience, mounted a vigorous opposition. Further complicating the negotiations was the reluctance of the provinces to accede any further control over wildlife management to the federal government.

Given the widely divergent positions of conservation groups and scientists on one side of the debate, and the provinces and business interests on the other, it took until 2002 for SARA to finally be passed. The federal government sought a middle ground, and this meant that many compromises were made. SARA ended up substantially weaker than its US counterpart. We will examine the specific strengths and weaknesses of SARA in Chapter 6.

While SARA was being developed at the federal level, many of the provinces adopted endangered species legislation of their own. By the time SARA was passed in 2002, eight provinces and territories had species at risk legislation in place, and five did not (Boyd 2003). The provincial legislation was generally weaker than SARA and featured the same compromises (see Chapter 3).

The War in the Woods

The advancement of conservation in the late twentieth century was not limited to the recovery of species at risk; it also included the management of landscapes. Landscape-based efforts began when rising environmental awareness in the 1970s led to demands for better management of industrial activity on public lands, most of which are forested.

Federal and provincial governments initially responded through commitments to manage forests for multiple values (such as wildlife), and not just timber supply. However, in practice, managers generally interpreted this directive to mean that other values were to be accommodated only to the extent that they did not significantly impinge on resource extraction (Wilson 1998). This did allow for some conservation gains, such as the protection of sites with low resource value. But fundamental changes in forest management were not forthcoming. Consequently, individuals and groups concerned about forests became progressively disillusioned with the government and their trust was eroded.

South of the border, forest management was also evolving, but along a different trajectory (MacCleery 2008). By the mid-1970s, studies had revealed that late-successional forests in the Pacific Northwest provided essential habitats for a suite of animal and plant species, including the northern spotted owl (Fig. 2.19). In response, conservation-minded scientists began to develop and promote new ecologically based approaches to forestry. These developments, together with a growing wilderness preservation movement, fuelled intense debate about the management of US public forests, most of which were under federal jurisdiction.

Fig. 2.19. A northern spotted owl. Credit: J. Hollingsworth.

The turning point came in March 1989, when federal district court judge William Dwyer issued an injunction on the harvest of virtually all national forest timber within the range of the northern spotted owl (i.e., most of the Pacific Northwest). He ordered the Forest Service to revise its standards and guidelines to ensure that the northern spotted owl remained viable, as required under the US *Endangered Species Act*.

When the dust finally settled, in the early 1990s, a new system of forest management, referred to as ecosystem management (see Chapter 7), had been adopted for all US national forests. Harvest volumes, which had been relatively consistent between 1960 and 1989, fell by over 80%, reflecting what the US Forest Service deemed necessary for maintaining the ecological integrity of national forests and the viability of species dependent on old-growth habitat (MacCleery 2008). These changes were backstopped by the US *Endangered Species Act*, which had no counterpart in Canada at the time. Nevertheless, Canadian conservationists were emboldened by the developments to the south and determined to see ecosystem management concepts applied here.

Another important development affecting the course of Canadian conservation was the release of *Our Common Future* (also known as the *Brundtland Report*) by the World Commission on Environment and Development in 1987 (WCED 1987). This high-profile report drew international attention to the importance of balancing economic and environmental objectives through **sustainable development**, which was defined as “development that meets the needs of the present without compromising the ability of future generations to meet their own needs” (WCED 1987, p. 43). The report also called for a tripling of the world’s protected areas to achieve adequate representation of all ecosystems. This recommendation formed the basis of the 12% protection target that was popularized in many countries, including Canada (see Chapter 8).

Our Common Future and the old-growth forest controversy in the US were elements of the broad resurgence of global environmentalism in the late 1980s that we encountered earlier in our discussion of species at risk legislation. In this milieu of heightened environmental salience, simmering discontent with forest management across Canada reached a flashpoint, resulting in the so-called "War in the Woods." During this period, the media once again displayed heightened sensitivity to environmental issues, and local stories that had previously lurked in obscurity were now cast onto the national and sometimes international stage.

Although the War in the Woods affected forests from coast to coast, BC was ground zero (Fig. 2.20). Most of the initial battles involved opposition to proposed harvesting in southern BC's last pristine watersheds, including South Moresby Island, the Stein Valley, and Clayoquot Sound. These early campaigns were primarily based on a wilderness preservation agenda, rather than a forest management agenda.

Fig. 2.20. Hundreds of individuals were arrested in protests against old-growth logging in BC in the early 1990s. Credit: R. Muirhead, Elphinstone Logging Focus.

Conservation groups advanced their forest protection objectives through broad networks of supporters and public outreach. The groups were adept at using symbolism and emotional appeal to win support for their cause, feeding into shifts in societal values. They were also highly effective in discrediting the forest industry's old-growth liquidation program and out-of-date harvesting practices. In later stages, the groups also used international public opinion and boycotts as leverage. The forest industry, for its part, spent millions of dollars on advertising campaigns, but public perceptions of the industry continued to decline despite these efforts (Wilson 1998).

In terms of public profile, the high-point of the BC campaigns occurred in the summer of 1993 when over 800 people were arrested for blocking logging trucks in Clayoquot Sound—the largest act of civil disobedience in Canadian history to that point in time. Television sets across the country beamed images of hundreds of people, from students to raging grannies, being dragged off to jail in defiance of an industry that had been the lifeblood of the BC's economy for almost a century. The protests did not result in immediate capitulation by the government, but most of the areas contested in the early campaigns were eventually protected.

The events in BC had ripple effects across the country, raising awareness and leading to forestry- related protests in many areas. The objectives and nature of the protests were different in each case. In Alberta, the trigger was the allocation, in 1987, of vast northern timberlands without public hearings, scientific study, or regional planning (Pratt and Urquhart 1994). In Ontario, the focal point was the proposed logging, in 1989, of the old-growth pine forest in the Temagami region, one the last of its kind in eastern North America. One of the protesters arrested in this case was Bob Rae, who would later serve as premier of Ontario. In Quebec, the film *L'Erreur Boréale*, directed by a popular folk singer, Richard Desjardins, generated public outrage over forestry practices in the province and demands for change. Forest protests even reached the east coast, as New Brunswickers battled to save the Christmas Mountains from harvest.

As the 1990s progressed, the place-based wilderness preservation agenda began to merge with the ecosystem management agenda imported from the US. A broad consensus emerged to protect 12% of Canada's lands and waters in sites that provided representation of all of Canada's natural regions. World Wildlife Fund Canada provided initial leadership through its ten-year Endangered Spaces campaign, launched in 1989 (Hummel 1989).

Several provinces initiated formal planning programs in the 1990s to complete or augment their parks systems, and efforts are still ongoing in some regions (Fig. 2.21). As of 2022, 12.6% of Canada's terrestrial area (land and freshwater) was protected, along with 9.1% of Canada's marine territory (ECCC 2022). Legislation governing parks was also strengthened during the late 1980s and 1990s. Of particular note was an amendment of the Canada *National Parks Act*, in 1988, which established that the first priority of national parks was to maintain or restore ecological integrity (GOC 2000).

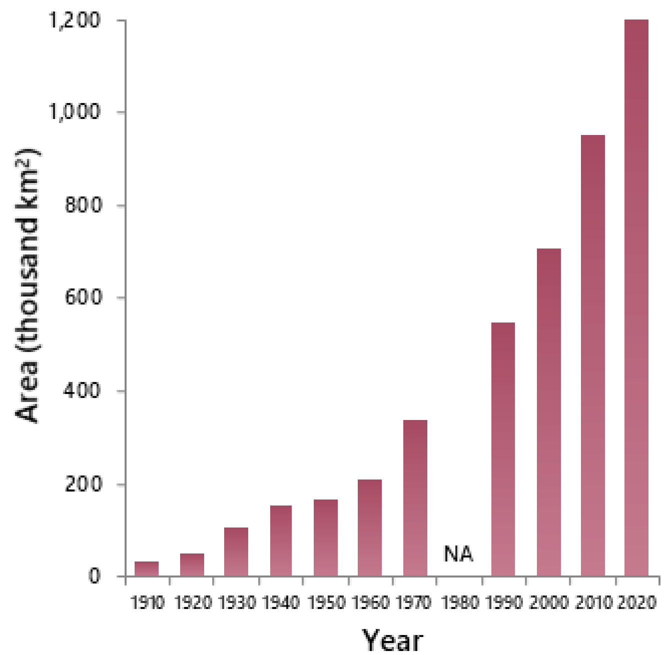


Fig. 2.21. The area of national and provincial protected areas in Canada, from 1911–2020. Source: SC 1983b and ECCC 2022.

The War in the Woods also led to changes in forest management which emphasized the maintenance of ecological integrity over the production of wood fibre. This shift was heralded by the *Canada Forest Accord*, signed by the Canadian Council of Forest Ministers in 1992 (CCFM 1992). As stated in the Accord, the goal of forest managers was to "maintain and enhance the long-term health of our forest ecosystems, for the benefit of all things both nationally and globally, while providing environmental, economic, social and cultural opportunities for the benefit of present and future generations" (CCFM 1992, p. 1).

Although federal, provincial, and territorial forestry ministers all signed the Accord, implementation was inconsistent across the country. The federal government could not enforce minimum standards or even ensure a coordinated response because authority over forest management rested with the provinces. The provinces blazed their own trails; some were progressive, and others were not.

BC and Ontario both passed legislation in 1994 that enshrined the goal of forest sustainability in law and set forth new requirements for forestry practices (GOBC 1994; GOO 1994). Both provinces also initiated land-use planning initiatives in the 1990s aimed at resolving broader conflicts related to land use. Forest legislation was also modernized during the 1990s in Saskatchewan, Quebec, Nova Scotia, and Newfoundland. The approaches varied but all included a commitment to forest sustainability and provisions for public participation (Boyd 2003). In contrast, Alberta, Manitoba, and New Brunswick made no effort to update their forestry legislation during this period.

The War in the Woods also disrupted the monopoly on decision making long held by government and industry. A large majority of the public now favoured forest protection over development and these values could no longer be marginalized (Lance et al. 2005). Furthermore, forest management was no longer a quiet, private affair. Conservation groups had expanded tremendously in terms of the number of members, financial resources, technical knowledge, and experience in communications. They, together with other engaged stakeholders (including Indigenous groups), were now a permanent fixture of the policy landscape and could not be sidelined. Though it was still a David and Goliath scenario with respect to financial resources and technical capacity, it was understood by all that conservationists were representing the conservation-minded public—like the part of an iceberg you see above the water line.

A related development was that some conservation groups, dissatisfied with years of half-hearted government responses, began to engage directly with forestry companies under the rubric of social licence (see Chapter 3). These efforts included direct negotiations over practices, the development of product certification schemes, and boycotts of selected high-profile companies. In some cases, these efforts proved to be quite effective.

For example, in Alberta, Alberta-Pacific Forest Industries became a lightning rod for popular discontent over forestry expansion in the late 1980s, making it the target of protests. This newly formed company emerged from its trial by fire with heightened environmental sensitivity. It became an early adopter of ecosystem management concepts coming from the US and quickly evolved into a vocal champion of progressive forestry, serving in the role the provincial government had abdicated (see Case Study 1, p. 259).

Because of these changing political dynamics, land-use decision making by the late 1990s was far more complex than it ever had been in the past (Luckert et al. 2011). The simple government-industry axis of information flow and decision making had evolved into a tangled web of interactions (Fig. 2.22). Although the large protests eventually subsided, governments, companies, and conservation groups continued to compete for the hearts and minds of the voting and consuming public in a "cold war" of claims and counterclaims about management successes and failures.

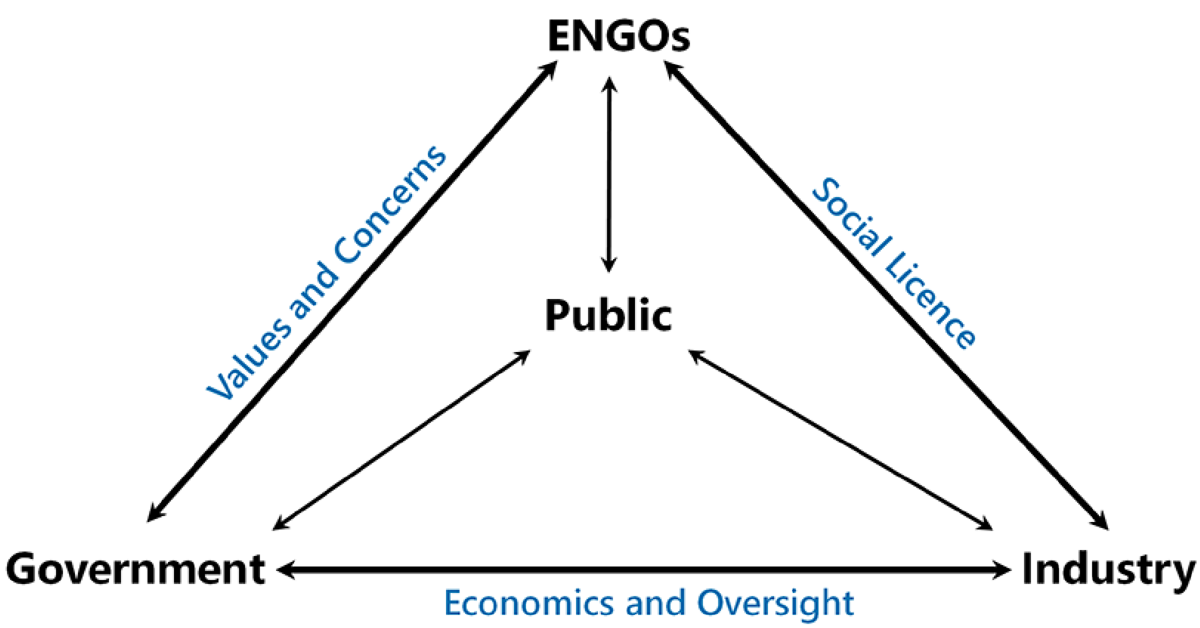


Fig. 2.22. A diagrammatic representation of the information flows characteristic of forest management decision making after 1990.

Unfortunately, the on-the-ground changes arising from the War in the Woods were much less impressive than might be expected given the grand commitments to forest sustainability made by governments and industry. In the US Pacific Northwest, maintaining the integrity of national forests meant reducing harvest levels by 80% (Mac-

Cleery 2008). In Canada, harvesting rates in the 1980s and 1990s did not fall at all; they actually increased (Fig. 2.8). Furthermore, late-successional forests generally remained primary targets for harvesting.

These differences reflect the simple fact that, in Canada, mill requirements continued to serve as the primary determinant of how much wood was cut. Even in BC, a leader of forestry reform, the Minister of Forests decreed that the average reduction in annual allowable cut resulting from the province's new forestry regulations would be no more than 6% (Wilson 1998). This defined, in no uncertain terms, the extent to which forestry reforms would be allowed to proceed. Harvest levels of forestry companies in other provinces were also maintained near their historical rates (Boyd 2003). As for lands taken out of production as protected areas, these were more than offset by new forestry allocations in other regions.

To be sure, several important changes did occur. Many ecologically important areas were protected during this period, including irreplaceable old-growth forests in southern BC. On the managed land base, though harvest rates did not decline, substantive improvements were made to harvesting practices. For example, progressive companies began varying the size and shape of cutblocks and leaving patches of live trees after harvest in an attempt to emulate natural disturbance processes (see Chapter 7). These efforts were guided by research undertaken by forestry companies, governments, and the academic community that sought to describe natural forest patterns and processes and to quantify the effects of human disturbances on forested ecosystems.

In summary, the War in the Woods was perhaps more evolutionary than revolutionary. But it did usher in a distinctly new era, featuring the actors, decision processes, and legacies that characterize forest management today.

CHAPTER III

THE SOCIAL AND POLITICAL DIMENSIONS OF CONSERVATION

The Social and Political Dimensions of Conservation

The General Public

Conservation is a broad, multilevel enterprise with both social and scientific components. In this chapter, we will focus on the social dimension, exploring the values, perspectives, and roles of the main participants. These participants include the public, environmental groups, the resource industry, Indigenous communities, and the government. In the final section, we will examine how government policies are developed and review the current conservation policy landscape. The role of conservation science and conservation practitioners will be discussed in the next chapter.

In contrast to many other countries, 90% of Canada's lands remain under public (Crown) ownership. This means the public has ultimate control over how most lands and resources are used, at least in principle. However, determining the wishes of the public, and sorting through the diversity of viewpoints, is far from easy. Some goals, such as the maintenance of biodiversity, clash with other goals, such as obtaining needed resources. Determining the best course of action in the face of such trade-offs is a central challenge for land and resource managers (Hauer et al. 2010; McShane et al. 2011).

The public is not only a landowner, but also a consumer. The opinions and preferences of the public are therefore of interest to companies that operate on public lands or make use of natural resources (Kennedy et al. 2009). Good alignment with public opinion may provide a boost to sales, whereas public dissatisfaction with a company's activities can result in the loss of markets or product boycotts.

Conservationists have a keen interest in public opinions and preferences as well because experience has shown that the success or failure of conservation initiatives often depends on the level of public support. Conservation groups may attempt to shape public opinion to generate support for their projects, but they also respond to public opinion by prioritizing initiatives based on the level of public interest.

Understanding Public Opinions and Values

Public opinion about conservation issues can be obtained through surveys, but interpreting the results and applying them in a decision-making context is challenging. Public opinion can be fickle, reflecting complex contingencies that are difficult to unravel (Tindall 2003). People may also hold viewpoints that are mutually inconsistent, often because the implications are not apparent or have not been thought through.

Consider the standard unprompted question that pollsters have been asking Canadians for several decades: What is the most important problem facing Canadians today? The environment is usually among the top four responses, but its salience tends to wax and wane in synchrony with the state of the economy (Environics 2012). In 2006 and 2007, with the economy doing well, Canadians identified the environment as the country's most pressing problem. One year later, as the effects of the 2008 recession took hold, concern for the environment was eclipsed by concern about the state of the economy and unemployment.

These findings may suggest that, when trade-offs are necessary, the economy trumps the environment in the public's mind. However, changing the context of the question yields a different result. When people are asked what the most important problem facing the country will be in the future, if nothing is done to address it, the environment is seen as significantly more important than the economy (Environics 2013). And when presented with specific conflicts between conservation and resource development, most Canadians favour conservation. For example, in a 2017 poll, only 25% of respondents agreed with the statement "Given the economic importance of the oil industry in Canada and the thousands of jobs it provides, it would make sense to continue with oil well development even if it meant that no greater sage grouse could survive in Canada" (IPSOS 2017, p. 3).

Because public opinion is so dependent on context it is difficult to obtain reliable insight from broad opinion surveys (Tindall 2003). Truly understanding what the public feels about a specific issue requires a combination of focused issue-specific polling, public forums, workshops, and working groups. Such consultations can be effective but are time-consuming and expensive, so they are generally reserved for high-profile issues.

Another approach for incorporating public input into decision making is to focus on values instead of opinions. **Values** are deeply held beliefs about what is desirable, right, and appropriate (McFarlane and Boxall 2000a; Tindall 2003). The benefit of working with values is that they are more stable than opinions and less contingent on external conditions.

The values associated with the environment and nature fall into two distinct categories: **utility values**, which relate to a human benefit, and **intrinsic values**, which concern nature alone. Examples of the main types of values within each of these two categories are shown in Table 3.1.

Table 3.1. Values of nature held by Canadians.

Value Type	Examples
Utility values	
Economic	Rents from the sale of resources, employment, food, tourism
Recreation	Bird watching, hiking, hunting, etc.
Ecosystem services	Nutrient cycling, water filtration; pollination, etc.
Aesthetic	Enjoyment and appreciation of the beauty of nature
Research/education	Learning about and understanding natural systems
Intrinsic values	
Moral/ethical	The right of species to exist and be valued for their own sake
Heritage	Passing on a healthy environment to next generation

Nature-related values are often described in economic terms. The direct economic benefits of resource extraction are easiest to quantify and are closely tracked by Statistics Canada. Canada ranks among the top five global producers of agricultural commodities, newsprint, lumber, oil and gas, aluminum, nickel, gold, potash, and diamonds. The resource sector, including agriculture, currently accounts for 23% of Canada's GDP and 24% of Canadian jobs (including indirect contributions; AAFC 2018; NRCAN 2018).

The economic benefits of non-industrial uses of nature, such as tourism, recreation, hunting, and fishing are also tracked. The *Canadian Nature Survey*, conducted periodically through a collaborative federal and provincial government effort, provides the most comprehensive data (CCRM 2014). In the most recent (2012) survey, 89% of adults said they participated in some form of nature-based activity during the year, and 57% took at least one trip of more than 20 km from their home to do so. Altogether, Canadians spend an estimated $41 billion on nature-related expenses per year, most of which is for transportation, accommodation, food, and equipment. Total expenditures are almost twice as high for non-motorized non-consumptive recreation as they are for motorized recreation and sport hunting combined, mainly because more people participate in non-motorized activities.

The ecological services that natural systems provide, such as water filtration and carbon storage, have historically received little attention, but that is now changing (Daily et al. 2009). The methodology for valuing these services generally relies on some form of replacement cost analysis. This entails estimating what it would cost society to replicate services that are provided for free by nature (but typically not appreciated). We will examine the ecosystem services concept in more detail in Chapter 4.

Attempts have also been made to place a dollar value on the intrinsic values of nature, using techniques such as "willingness-to-pay" surveys (Rudd et al. 2016). However, the results have not been compelling. Such studies have been criticized for being unrealistic and generating findings that are not repeatable (Nunes and van den Bergh 2001; Spangenberg and Settele 2010; Chan et al. 2012). This presents a problem because, when it comes to conservation, the moral, aesthetic, and heritage values of nature (Table 3.1) are often the primary drivers of public opinion and action. The thousands of people that took to the streets across Canada in the 1990s in protests over logging were not motivated by forest recreational values or ecosystem services. They valued nature for itself and wanted to see it protected.

Insights from Social Psychology

An alternative approach to understanding public values and opinions is provided by social psychology. Whereas economists seek to express public values in terms of a common unit of measure (dollars), social psychologists seek to characterize the range of viewpoints that exist and understand the causes of this diversity (McFarlane and Boxall 2000b).

Research by social psychologists suggests that most Canadians hold the nature-related values listed in Table 3.1 to some degree but differ in the relative importance ascribed to each. Value weightings tend to cluster in predictable ways, resulting in relatively stable and internally consistent conservation worldviews. These worldviews exist on a spectrum from **anthropocentric** (human-centred), in which utilitarian values are seen as most important, to **biocentric** (nature-centred), in which nature's intrinsic values predominate (McFarlane and Boxall 2000a; Tindall 2003).

Among the general public, an intermediate conservation orientation is most common, implying a shift from earlier periods when utilitarian values dominated (Wagner et al. 1998; McFarlane and Boxall 2000b; Kennedy et al. 2009). Individuals with this perspective support resource extraction but will not tolerate permanent damage to the environment. There is a strong expectation of sound ecological management and a suspicion that it may not be happening. There is also strong support for creating additional protected areas (IPSOS 2017). When asked to choose between protecting jobs and protecting the environment, this group typically sides with the environment (McFarlane and Boxall 2000b; AFPA 2006; IPSOS 2017).

The public also includes individuals with more extreme views. Those with a strong biocentric orientation tend to find current management practices inconsistent with their values and favour a much more protectionist approach (McFarlane and Hunt 2006). Those with a strong anthropocentric orientation, which is least common, usually consider current management to be adequate for protecting the environment or may even feel that existing regulations are too onerous.

Personal value orientations shape attitudes toward specific issues. But attitudes also depend on awareness and knowledge about the issues (McFarlane and Boxall 2000a). In the case of biodiversity conservation, this knowledge is largely gained second-hand, because most Canadians live in cities and towns far removed from the natural landscapes that are being threatened. Organizations that control information, including the government, environmental groups, industry, and the media are therefore able to set agendas (i.e., focus attention on some issues over others) and influence opinions.

Clearcut forest harvesting provides an illustration. A person with an intermediate conservation orientation is likely to support the general idea of harvesting trees but will expect that it is done in an ecologically sustainable manner. Industry sources may inform her that clearcut harvesting is ecologically benign because it mimics natural disturbances and facilitates effective regeneration. Environmental groups may inform her that clearcutting is ecologically damaging because it simplifies forest structure and leads to the progressive loss of old-growth forest habitat. Her ultimate support or rejection of clearcutting may therefore depend, not on her conservation orientation, but on where she obtains information and the level of trust she places in different sources. Scientists and environmental groups are usually afforded a higher level of trust than government and industry (Fig. 3.1), which partially explains why public perceptions of resource development are often negative.

Attitudes toward conservation issues are also influenced by socio-economic factors (McFarlane and Boxall 2000a). The most important factor is place of residence: urban or rural. Canadian society is now highly urbanized; 69.1% of the country's population lives in just 33 metropolitan areas, most of which are located within 100 km of the US border (SC 2015). For most urbanites, the value of resource development is as abstract as the intrinsic value of nature. For these individuals, a decision to favour protection over development is easy to make because there is little direct impact on their lives (at least nothing that is perceptible).

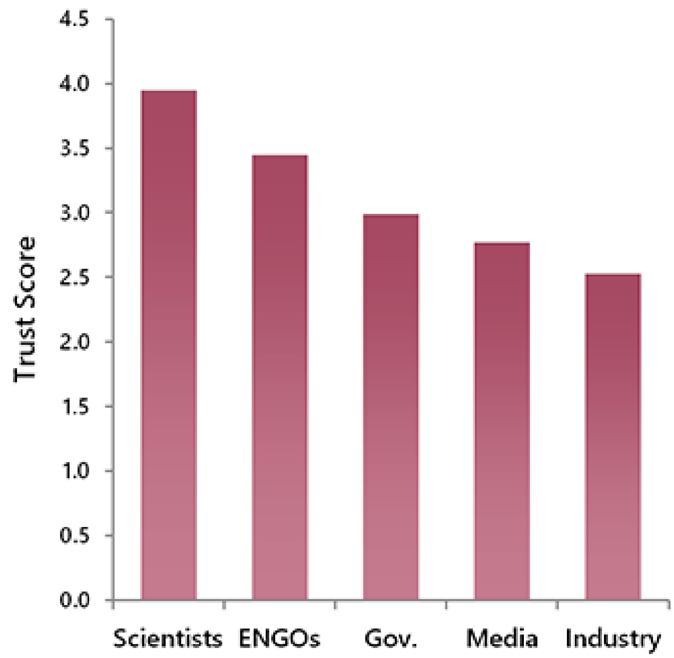


Fig. 3.1. The level of trust by the public in various information sources. Source: Lang and Hallman 2005.

Canadians who live on the land or in small towns and villages see things differently. For many rural communities, resource extraction is the foundation of the local economy. So the economic benefits of natural resources are understandably more important for them than for most urbanites. Rural individuals are also more knowledgeable about land-use issues than urbanites, more engaged, and expect a greater role in management decision making (Parkins et al. 2001; McFarlane et al. 2007). They also tend to draw on industry for information about land-use issues, whereas urbanites rarely do (Parkins et al. 2001). Frustration with decisions made by far-off city dwellers can run high because rural people are usually the ones that end up bearing the burden of environmental protection measures.

Despite their dependence on resource development, it would be a mistake to assume that rural residents have a disregard for the broader values of nature. Concern with maintaining environmental health is quite high among many rural dwellers, though they are likely to prefer approaches that involve careful management over strict protection (McFarlane and Boxall 2000b; Huddart-Kennedy et al. 2009; McFarlane et al. 2011). It is also worth noting that rural residents are not all of one mind. Individuals with a strong biocentric orientation may not be as common as in urban areas, but they do exist. And many rural communities are experiencing a "greening" effect as a result of an influx of city dwellers seeking a rural lifestyle (Huddart-Kennedy et al. 2009).

To summarize, the public is not a monolithic entity. Most Canadians have a moderate to strong biocentric orientation, but a sizeable minority place greater emphasis on utility values. In either case, the public expects, and wants to believe, that the government will manage natural resources on their behalf in a way that reflects and respects their values. The implication is that resource managers must acknowledge and address the full range of values outlined in Table 3.1, not just the ones that are easily quantified. Because some values conflict with each other, compromise will usually be necessary. The public can accept that but expects the decisions to be fair and balanced.

Environmentalists

Who are the Environmentalists?

To those who oppose their views, environmentalists are often characterized as “special interests,” somehow distinct from the rest of society. This simplistic characterization is at odds with what we know about public perspectives. Conservation worldviews exist along a continuum, and environmentalists are simply individuals drawn from the biocentric end of this spectrum (McFarlane and Boxall 2000b; McFarlane and Boxall 2003). There is no distinct “us” and “them.”

Environmentalists are themselves a heterogeneous group. The main differentiating factor is the level of engagement. Those on the cusp of environmentalism might come out to a nature-related event, donate some money, or sign a petition if asked. They may also make purchasing decisions based on their perception of environmental impact (CCRM 2014). Such individuals number in the millions in Canada.

The next level of engagement is typically membership in an environmental non-government organization (ENGO). These individuals gain greater awareness about issues through group newsletters and websites, and they support the aims of their organizations by providing funding and perhaps soft advocacy such as letter writing. Further engagement often involves volunteering. Some ENGOs depend on volunteers to help implement programs that are planned and organized by staff. Many smaller groups are entirely run by volunteers.

The highest level of engagement involves serving on the board of an ENGO or working as paid staff. Outside of Ducks Unlimited Canada and the land trusts, which have relatively large staffs, the full-time paid staff of conservation-oriented ENGOs in Canada is a small select group, collectively numbering in the hundreds (Grandy 2013). These are highly dedicated individuals, and for most of them, working for a conservation organization is as much a mission as it is a job.

Increased engagement is accompanied by increased knowledge about conservation issues and often involves a hardening of opinions concerning resource development (McFarlane and Boxall 2003). A new volunteer representing a local group at a decision-making forum may only be able to speak in terms of basic value sets. More experienced volunteers may have acquired considerable knowledge on issues they have worked on for many years. Paid staff are likely to have formal training in ecology, law, or management and will typically have a depth of technical expertise on issues under discussion that matches or exceeds that of government representatives and other stakeholders. This expertise is gained within the context of a strong biocentric worldview and will therefore be subject to certain filters. An environmentalist and a forester may both agree on the need for sustainability but hold completely different perspectives on what that looks like in practice.

From this description, it is apparent that the environmental community has a pyramidal structure (Fig. 3.2). A potential concern with this organizational structure is that decisions concerning objectives, tactics, and the allocation of effort are usually made by a small number of people, including paid staff and active board members (Brulle and Jenkins 2010). Critics have argued that because of this disconnect between group leaders and the organization's membership, ENGOs really are a special interest after all.

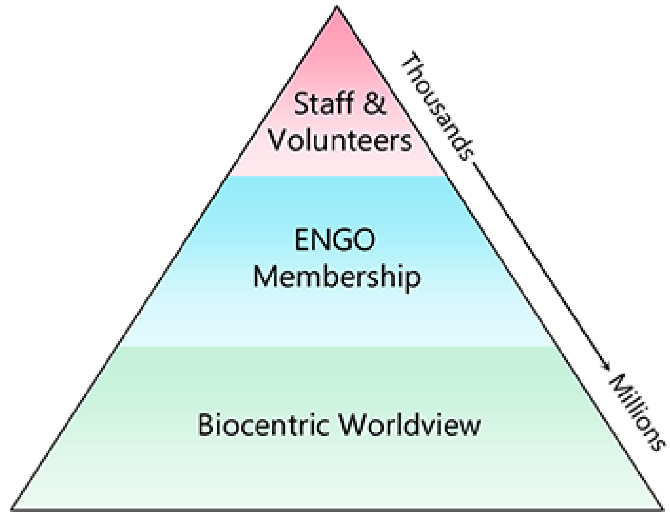


Fig. 3.2. The pyramidal organization of Canadian environmentalists.

What these critical views fail to consider is that membership and support of ENGOs is a voluntary process and there are many groups to choose from. If a group is ineffective or chooses to pursue a narrow or unpopular agenda, members vote with their feet, taking their support with them. This process ensures that ENGOs and their leaders remain aligned with the interests of their members. Without a strong membership, groups lose their core funding and, just as important, the political power that comes with representing the public interest.

Another potential concern is that many ENGOs receive a substantial proportion of their funding from large philanthropic foundations and the government (Tedesco 2015). Because of this, critics have charged that environmental groups may be more attuned to the interests of external funders—another special interest—than their members (Krause 2013).

Foundation and government funding certainly comes with strings attached; however, the relationship is not coercive. ENGOs seek external funding to execute programs they have developed on the basis of carefully crafted strategic plans. Strategic plans are in turn a reflection of a group's mission and vision, which define what it stands for, what it does, and where it fits in the broader ENGO ecosystem. A group's mission and reputation will not readily be jeopardized for individual funders, which come and go over the years. Nevertheless, external funding does change the opportunity landscape and can thereby influence which projects advance and which remain on hold.

Foundations are best characterized as investors seeking the highest rate of return. Smaller conservation groups working on local issues receive some attention, and the occasional long-shot is entertained. But the bulk of the funding is directed to programs likely to have a broad impact and to organizations with the size and technical expertise needed to make significant progress (Brulle and Jenkins 2010).

In summary, much goes on behind the scenes that determines what individual ENGOs do, and membership is rarely involved in making these decisions. In this respect, ENGOs are no different from most large member-based organizations and elected governments. ENGO members express their support by renewing their membership each year and providing ongoing financial support. This base of support is very broad in Canada. According to the 2012 *Canadian Nature Survey*, 4.8 million Canadians annually donate money to nature conservation organizations (CCRM 2014). Active membership in these organizations is collectively in the hundreds of thousands (Grandy 2013).

Support for ENGOs is also reflected in public opinion about who should be allowed to influence decision making related to resource management. The groups that typically receive the highest support are scientists, ENGOs, and local communities (Robson et al. 2000; Parkins et al. 2001; McFarlane et al. 2007). Support for ENGO inclusion is highest among urban dwellers, but even among rural residents, support is still substantial (Fig. 3.3).

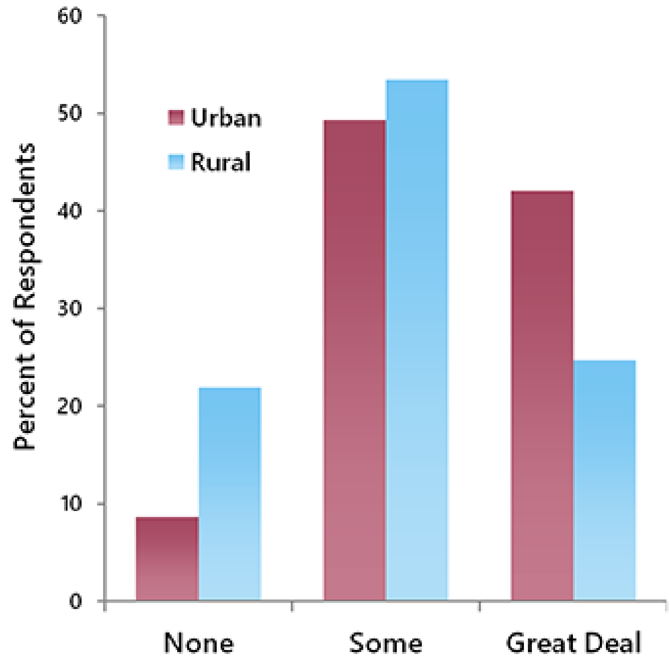


Fig. 3.3. Survey responses to the question: How much influence should ENGOs have in forest management decision making? Source: McFarlane et al. 2007.

The ENGO Ecosystem

Individual environmentalists are typically concerned about a wide range of issues, from biodiversity to pollution to global warming. Environmental groups, however, tend to specialize. In the following overview, we will explore the diversity that exists, focusing on groups that are active in biodiversity conservation to at least some degree.

The vast majority of conservation-oriented ENGOs are small volunteer-run organizations that work on local or regional issues (Hall et al. 2005). These include stewardship groups, watershed councils, naturalist clubs, and a host of "friends of …" societies. There are also many organizations that form in opposition to proposed developments. These groups are generally composed of local citizens that share an interest in some aspect of conservation and have joined together to do what they can to help. Collectively, these groups make a substantial contribution to conservation by raising awareness of local issues and influencing resource management at the local level. These are also the groups that regional and provincial-level planners turn to for obtaining local perspectives on land use.

On the other end of the spectrum are the large provincial and national conservation organizations. Though few in number, these groups have the largest memberships and receive most of the available funding (Hall et al. 2005). Because of regional overlap, these large ENGOs must compete for members, funding, and attention to their ideas. Consequently, they have differentiated to fill specific niches (Table 3.2). Some are confrontational; others are cooperative. Some are generalists, and others are specialists. Some are grassroots organizations with regional chapters, whereas others are run more like centralized corporations. At the most fundamental level, the groups can be divided into those that primarily conduct hands-on conservation and those that operate mainly in the public sphere, conveying information and ideas.

Table 3.2. Overview of national ENGOs that engage in conservation activities in Canada, ranked by annual revenue.[1]

Organization	Revenue (millions)	Staff[2]	Origin	Focus
Nature Conservancy of Canada[3]	185.2	514	1962	Land trust
Ducks Unlimited Canada[3]	104.6	486	1938	Wetland protection
Greenpeace[3]	47.6		1971	Multi-faceted
Canadian Wildlife Federation	33	65	1962	Wildlife conservation
World Wildlife Fund Canada[3]	30.7	110	1967	Wildlife conservation
David Suzuki Foundation	14.7	83	1990	Multi-faceted
Canadian Parks & Wilderness	11.7	92	1963	Wilderness preservation
Birds Canada	8.8	119	1967	Wildlife research
Ecojustice	8.3	74	1990	Environmental law
Nature Canada	6.9	56	1939	Naturalist clubs
Wildlife Conservation Society[3]	6.5	52	2004	Wildlife research
Environmental Defence	4.5	54	1984	Multi-faceted
Stand.earth[3]	3.7		2000	Multi-faceted
Wilderness Committee	3.3	26	1980	Wilderness preservation
Wildlife Preservation Canada	1.4	44	1985	Wildlife conservation
Trout Unlimited	1.3	16	1972	Stream protection
Sierra Club Canada (national)	1.2	59	1969	Multi-faceted

[1]Data obtained from Canada Revenue Agency charity listings, downloaded Jan. 2023 from https://www.canada.ca/en/services/taxes/charities.html?request_locale=en.
[2]Staff includes both full- and part-time employees.
[3]For international groups, the information provided here refers to Canadian operations only.

The groups that engage in hands-on conservation rarely generate news headlines, but they account for the lion's share of Canadian conservation staff and expenditures (Table 3.2). Most of these groups are focused on habitat preservation and restoration, as exemplified by Ducks Unlimited Canada, the Nature Conservancy of Canada, and other land trusts. They work cooperatively with government and industry and derive a large portion of their funding from these sources.

These groups protect land mainly through partnerships with private landowners in agricultural regions. The priorities for protection are identified in strategic plans that reflect conservation need, opportunity, and the group's main interests (e.g., wetlands). Once willing partners within priority areas are identified, lands are secured using three main approaches: land donations, land purchases, and conservation easements (see Chapter 8). These groups also promote conservation through their government and industry relationships and they often participate in government planning initiatives.

Fig. 3.4. A Ducks Unlimited wetland restoration project. Credit: Department of Natural Resources, Wisconsin.

In addition to securing land for protection, some of these groups engage in ecological restoration. For example, they may re-establish natural vegetation on cultivated lands or restore the hydrological integrity of degraded wetlands (Fig. 3.4). Land trusts usually focus their restoration efforts on land parcels that they have protected; however, Ducks Unlimited conducts its wetland restoration projects on a much broader scale.

The other ENGOs—those that do not engage in hands-on conservation—operate in the public sphere, rather than with individual landowners. One of several methods they use to advance conservation is public education. Education raises the profile of biodiversity among the public and helps individuals maintain a connection to nature and its values. ENGOs organize nature walks, give public presentations, generate media stories, and prepare educational materials for use in schools among other activities.

ENGOs also serve in a watchdog role, providing eyes and ears for the public on issues related to conservation. ENGOs have extensive networks that connect them with what is happening on the land. They also draw heavily on research from the scientific community, with which they have a symbiotic relationship (see Chapter 4). Through this process, existing and emergent threats to biodiversity are identified, and priorities for action are brought to the public's attention. In addition, ENGOs cast light into shadowy areas of policy, exposing the full costs and risks of development and forcing governments to defend their decisions. They also hold governments and industry to account when policies and regulations are not implemented or properly enforced.

Lastly, ENGOs serve as advocates, providing a voice for nature and the conservation-minded public in decision-making forums. ENGOs also introduce new ideas into the public discourse and promote solutions to conservation problems in the form of alternative management approaches. These are critical functions because no other stakeholders effectively represent the biocentric views of Canadian society. For example, industry advisory groups tend to have more of an anthropocentric worldview (Fig. 3.5). So do resource professionals.

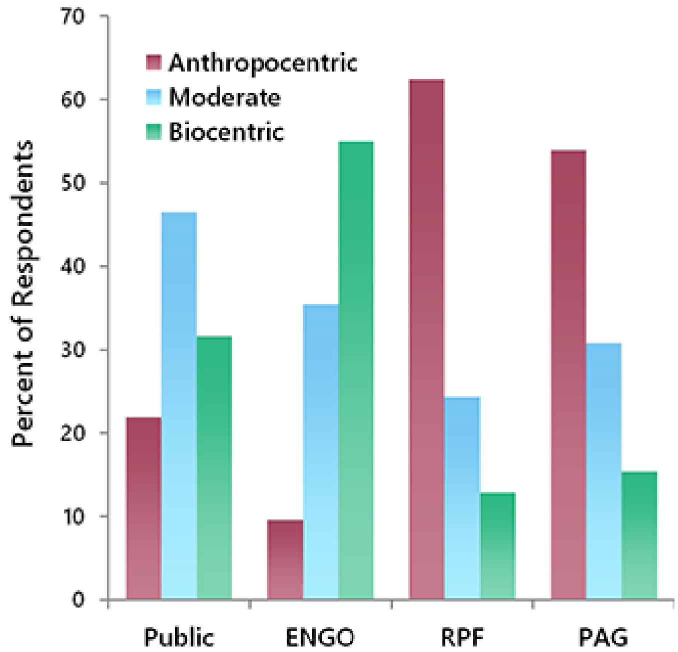


Fig. 3.5. The distribution of conservation orientation among stakeholder groups. ENGO = members of an environmental group; RPF = registered professional foresters; PAG = members of a public advisory group. Source: McFarlane and Boxall 2000b.

Advocacy can take many forms. It can be reactive, such as responding to proposed projects, policies, and legislation (Fig. 3.6). It can also involve proactive lobbying of politicians and government bureaucrats in support of ENGO programs and campaigns. Such lobbying has historically been highly constrained by tax laws, but these restrictions were loosened in 2018. Much advocacy also gets done through the creative use of educational materials and the support of volunteers. Finally, advocacy efforts are sometimes aimed directly at resource companies, using both carrot and stick approaches (discussed below).

Fig. 3.6. Drawing attention to conservation issues through protests and demonstrations remains an important tool for environmental groups, such as Elphinstone Logging Focus, shown here protesting the logging of old-growth forest in BC. Credit: R. Muirhead.

Setting Priorities

Conservation-oriented ENGOs share a common interest in maintaining biodiversity. However, individual groups lack the capacity to address this goal in its entirety, so they concentrate their efforts on specific issues. In so doing, the groups play a major role in setting the conservation agenda within the entire public sphere.

Priority setting by conservation groups typically begins with a regional assessment of threats, often with an emphasis on human activities that disturb habitat. The level of threat is presumed to increase with the intensity of disturbance and its extent. Scientific information also feeds into the process. Reports about the effects of different types of human activities on ecological systems and species at risk are of particular interest.

The determination of conservation priorities also includes an assessment of opportunities, barriers, and the overall likelihood of a project's success (Brulle and Jenkins 2010; Dart 2010). Given limited capacity, conservation groups need to focus their activities where they will do the most good. But this determination is far from easy given the basic paradox of conservation: protection is easiest to achieve where it is least needed. Groups must often choose between projects that address a critical conservation need, but come with high barriers to success, and projects of lower importance that are more achievable. Such determinations are all the more difficult because the likelihood of success is difficult to predict.

The main factors that ENGOs consider when judging the likelihood of success include:

- **Political opportunities and barriers.** Political receptivity to conservation waxes and wanes over time. Therefore, prioritization often includes an element of opportunism, as ENGOs make the most of opportunities that occasionally present themselves on specific issues. Conversely, conservation issues that face strong political opposition may be assigned a lower priority.
- **Public support.** ENGOs derive political influence from their claim to represent the biocentric views held by a broad segment of society. The more evidence there exists of this linkage, the more influence they have. This means that ENGOs must carefully consider projects in terms of their public appeal. All else being equal, it is much easier to generate broad public support for the protection of a majestic old-growth forest or a charismatic mammal than it is for a mosquito-infested peat bog or an endangered lichen. Despite ENGO aspirations of conserving biodiversity as a whole, such realities cannot be ignored.
- **Allies**. Profile, capacity, and influence in political decision making can be strengthened through alliances with other stakeholders. Therefore, effort may be channelled into projects that align with the interests of other organizations, such as other ENGOs, Indigenous communities, progressive resource companies, and funding foundations. Collaboration with other ENGOs works best when the groups involved have complementary approaches, but tends to be avoided when there is a potential for redundancy. For example, confrontational groups may use high-profile campaigns to raise awareness of issues, creating space for local policy-oriented groups to advance specific solutions.
- **Opposing values.** An important determinant of success is the degree of conflict with competing values. Species and ecosystems with the poor fortune of existing in areas of high resource value are difficult to protect. A modifying factor is the extent to which solutions exist for reducing or eliminating the main points of conflict, allowing for win-win outcomes.
- **Strategic value.** Though the likelihood of success in the short term is an important consideration, projects may also be undertaken for their strategic value in advancing conservation over longer timeframes. Favourable political environments do not arise spontaneously. They are usually the result of long-term efforts that may offer little apparent success in their early stages.

Priority is influenced not only by the likelihood of success but by the ability to measure success (Grandy 2013). This is one reason why habitat protection efforts tend to be favoured: the outcomes are clear and robust. Protected areas established through conservation campaigns in the twentieth century continue to protect habitat today. The same goes for private land that was purchased or deeded for the purpose of conservation. There is very little ambiguity about what was achieved or who achieved it.

Measuring success is much more challenging for projects related to conservation policies and practices. In the policy realm, many players are involved, and everything is interconnected, so it is difficult to have a decisive effect. And when change does occur, it is hard to know who was responsible for what. Moreover, experience has also shown that success in this area is often illusory. Hard-won policies may fail to be implemented, and commitments may be repealed with a change in government.

Lastly, priorities are influenced by institutional factors, such as a group's primary mission, technical expertise, and contact networks. Spatial scope also plays a role. Provincial and regional groups often engage in local issues that national groups overlook or choose to avoid because better prospects can be found elsewhere.

In summary, the projects that ENGOs undertake, and by extension, the conservation issues that reach the public consciousness, are not simply a reflection of conservation need. ENGOs usually approach conservation from a highly practical perspective, blending ecological science and social factors when setting priorities. The process is messy and subject to various shortcomings, but it reveals important conservation realities. Conservation strategies and plans have no value if they are incapable of effecting change. There is a need for plans that offer solutions that are workable in a world of competing values and that resonate with the broad public. In Chapter 4, we will see that conservation practitioners working in scientific and management capacities face very similar challenges.

Industry

Resource development is associated with utilitarian values and has several distinct forms of support. The resource industry is a provider of necessary goods and an important contributor to employment and the general health of the economy. This generates support across a broad swath of society, including individuals with a moderately biocentric worldview. Such support is generally accompanied by an expectation of sound environmental management.

Another form of support resides in rural communities that rely on resource companies for employment, tax revenue, and opportunities for local businesses. Support is typically highest among company staff; however, working for a company does not translate into unconditional support for what the company does. Individuals may work for a company simply because it offers a high wage or because few alternatives for employment exist in small communities.

A third group of supporters, and arguably the most important from a company's perspective, are shareholders. Resource companies may be valued by the public for the raw materials, jobs, and taxes they provide, but this is not why they exist. Companies are business enterprises that exist to generate a financial return on capital provided by their owners. This distinction is important because the underlying profit motive directs and constrains company decision making. Company directors have a fiduciary duty to their shareholders to use the company's assets efficiently and effectively, which in practice means optimizing production and minimizing costs. Biodiversity conservation generally enters this equation as a cost, so decisions supporting protection measures must be justified by a business case, such as risk management related to social licence (discussed below).

In land-use decision making, resource companies no longer enjoy a monopoly on government attention, but they still retain important structural advantages (Wood et al. 2010). In many cases, they are the only stakeholder that contributes significantly to the local and regional economy. This affords them considerable influence with government decision makers, who are responsible for both land management and fiscal management. Governments must also honour the rights that companies have been awarded under tenure and licence agreements. The alignment of resource companies with the interests of local communities further boosts their influence. Finally, resource companies have a high level of government access through personal networks that are built, in part, on the reciprocal flow of staff between government and industry (Meghani and Kuzma 2011). Through these processes, industry is able to exert considerable concentrated political power at the local level, whereas the power of ENGOs is mostly urban based and less focused.

Industry and the Environment

Approvals for industrial projects normally come with environmental constraints. Companies must also respect the laws and policies of the jurisdiction in which they operate. Although companies are occasionally caught breaking the rules, they generally accept the imposed environmental constraints as one of the costs of doing business.

A challenge companies face is that environmental constraints are not fixed. Sometimes new information comes to light, such as better understanding of the ecological effects of an industrial practice. Or it may be that long-term cumulative impacts have pushed a local ecosystem or species to a critical state. In many cases, the impetus for change comes from evolving social concerns and priorities. As we saw in Chapter 2, public values and environmental awareness underwent a seismic shift in the late twentieth century, leading to demands for higher standards of environmental protection. Canada's resource sector responded with broad commitments to environmental sustainability (Fig. 3.7; Hilson and Murck 2000; Lazar 2003).


ENVIRONMENT POLICY

WE ARE COMMITTED TO RESPONSIBLE STEWARDSHIP OF THE ENVIRONMENT THROUGHOUT OUR OPERATIONS

WE WILL:

- Comply with or exceed legal requirements.
- Comply with other environmental requirements to which the company is committed.
- Achieve and maintain sustainable forest management.
- Set and review objectives and targets to prevent pollution and to continually improve our sustainable forest management and environmental performance.
- Provide opportunities for interested parties to have input into our sustainable forest management planning activities.
- Promote environmental awareness throughout our operations.
- Conduct regular audits of our forest and environmental management systems.
- Communicate our sustainable forest management and environmental performance to our Board of Directors, shareholders, employees, customers and other interested parties.

JANUARY 1999

JOHN MACMILLAN
President and Chief Executive Officer

DON SMITH
Chairman

FOREST PRODUCTS LTD.

Fig. 3.7. Many resource companies have developed policies affirming their commitment to environmental protection.

Environmental sustainability has implied tests and standards that often exceed government regulations. This extra-governmental standard has been called the "**social licence to operate**," and presents an ongoing vulnerability (Boutilier 2014). Industry has done much in recent years to improve its practices and can point to many success stories, but the undeniable reality is that Canada's biodiversity is in a worse state today than when the resource industry's commitments to sustainability were made (CCRM 2010; EC 2012a). In the context of sustainability, doing better is not the same as doing enough.

One approach companies use to reduce their environmental risk profile is continuous improvement of their practices. In some cases, innovative solutions can be found that support biodiversity without being disruptive or costly to implement. For example, in forested areas, seismic exploration for oil and gas used to require 6–8 m-wide linear corridors created by bulldozers, and a large proportion of these lines failed to regenerate in subsequent decades (Lee and Boutin 2006). Today, equipment is available for conducting seismic exploration on lines that are only 2.5 m wide, and the cost of the new equipment is offset by reduced penalties for timber damage.

In most cases, new conservation measures impact company profits to some degree. A natural tendency is for companies to implement the least costly practices first and to delay or avoid implementing more costly practices. For example, in the forestry sector, efforts to maintain natural forest structure have focused on varying the size and shape of harvest blocks. Some cost is involved because planning is more complicated and staff must be retrained, but these costs are manageable. Other, more costly measures, such as retaining old-growth forest, have typically been resisted (see Chapter 7).

Companies also reduce their environmental risk profile through public outreach. Their aim is to convince people that their operations are ecologically sustainable and further protection measures are unnecessary (Fig. 3.8). In this, companies have several factors operating in their favour. Their efforts over the years to improve practices legitimately bolster their claim of sustainability. This is a message that resonates well with a large segment of the public because people would like to believe that their use of natural resources is not contributing to the deterioration of the environment. Less constructively, companies may use obfuscation as a tool for masking problematic issues. The complexity of conservation issues makes this an effective tactic (Jacques et al. 2008). Finally, resource companies have operating budgets that can support large, sustained public relations efforts that reach a wide audience. Money is also often used to buy goodwill; for example, by funding the purchase of land for conservation and by supporting local community initiatives.

Fig. 3.8. An example of the "green" advertising that some resource companies use to promote the environmental friendliness of their products.

Working against the resource companies is a low level of public trust (Fig. 3.1). In an international survey, the four industrial sectors with the lowest trust ratings were mining, chemical, oil, and tobacco (Boutilier 2014). Mistrust arises from periodic high-profile disasters that undermine industry claims of reliable management, such as the catastrophic tailings pond failure at the Mount Polley mine in BC in 2014 (GOBC 2014). Trust is further eroded by the all-too-common exposure of fraudulent claims. If a highly reputable company like Volkswagen is willing to manipulate their emissions data in a multi-billion-dollar gamble to improve sales (Neil 2015), it makes people wonder what other companies may be hiding.

The other major challenge industry has in defending its environmental claims comes from ENGOs. In contrast to the public, ENGOs have the technical capacity to sort truth from fiction when it comes to claims about sustainability. Thus, industry efforts to simply relabel old practices with new terminology are usually unmasked. ENGOs do not have the budgets to match industry's public relations efforts, but they do enjoy a high level of public trust, which works in their favour. For its part, industry is often able to successfully counter ENGO demands for change with arguments based on scientific uncertainty and the need for more research. In some cases, additional study is indeed the most appropriate course of action. But all too often, protracted debate and concerns about uncertainty are used simply as tactics to delay action (Aklin and Urpelainen 2014).

Market Forces

The extent to which companies will implement progressive environmental practices, in excess of legal requirements, boils down to a business decision. In some cases, the logic is to avoid having even more restrictive regulations imposed by governments seeking to address unresolved environmental concerns. In many other cases, the costs and benefits of progressive practices are framed in terms of market access, which is another way of expressing social licence. According to the *Canadian Nature Survey* (CCRM 2014, p. 1), 57% of Canadians purchase "products and services that are more environmentally friendly than their competitors." For companies, the potential gains and losses associated with such purchasing decisions must be weighed against the costs involved in raising the bar of sustainability.

In practice, the linkage between consumers and resource companies is highly convoluted. Consumers generally do not buy raw materials like crude oil and copper ore; they buy manufactured goods like gasoline and extension cords. Therefore, to establish a business case for conservation, there must be a way of tracking the supply chain, so that customers can identify end products made from sustainably derived raw materials. Also, claims concerning sustainability must be substantiated through an objective and impartial assessment of operating practices against meaningful standards. Without this step, companies that implement progressive practices cannot differentiate themselves from competitors that pay only lip service to sustainability. Established standards also help protect consumers from being misled.

Tracking the chain of custody is very difficult within the oil and gas and mining sectors, with the notable exception of diamonds. Therefore, the most progress in product certification and tracking has occurred in the forestry sector. Forest certification was pioneered by the Forest Stewardship Council (FSC) in the 1990s, and to date, more than 2 million km^2 of forest have been certified under this system, in 86 countries (Fig. 3.9). In Canada, 22% of managed forests are now FSC certified (FSC 2020).

The way FSC certification works is that companies request (and pay for) an audit of their operations by an accredited certifier. To gain certification, companies must meet FSC's environmental and social standards, developed by a group of ENGOs, retailers, unions, and Indigenous peoples (FSC 2020). FSC also runs a chain-of-custody system that tracks fibre from certified forests through the supply chain all the way to the customer. Thus, the overall network comprises not only the FSC organization, but industry, ENGOs, certification agencies, wood product manufacturers, and retailers.

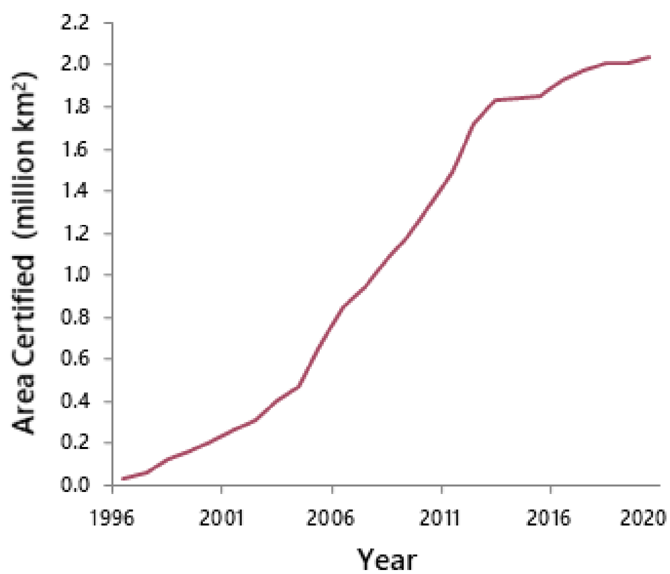


Fig. 3.9. The area of forest under FSC sustainable management (1996–2020). Source: FSC 2020.

In practice, most of the demand for FSC-certified products comes from manufacturers and retailers, rather than directly from consumers (Schepers 2010). Like primary producers, manufacturers and retailers seek to differentiate themselves in the marketplace, and many choose to do so by becoming "green." A good example is IKEA, which has established itself as a leader in sustainability. It has developed a sustainability plan that includes criteria and targets across the full breadth of its business enterprise. For its wood products, the company's objective is to purchase at least 50% of its supplies from FSC-certified or recycled wood sources (IKEA 2014).

Another certification system in widespread use is run by the Marine Stewardship Council (MSC 2017). Like FSC, the Marine Stewardship Council has established sustainability standards, in this case for marine fisheries. It also oversees producer certification, runs a chain-of-custody system, and tracks the effects of certification. A key difference with Marine Stewardship Council certification is that it is less dependent on intermediaries. The main target is the purchase of fish at the grocery store by individual consumers.

Certification is also in place for the producers of organic agricultural products. In this case, the system, dubbed the Canada Organic Regime, is overseen by the Canadian Food Inspection Agency and is regulated under federal law. The federal government's involvement was largely driven by European requirements related to agricultural imports. Provinces have their own systems for organic food produced and sold in the same province. Organic food standards are focused on pesticide use and soil health and make no direct mention of maintaining native biodiversity; however, they do benefit biodiversity indirectly (GOC 2018).

Although product certification does lead to improved practices (Moore et al. 2012), it is not a panacea. A fundamental limitation is that the criteria for sustainability must be achievable (Schepers 2010). If the bar is set too high, the cost of implementation will exceed the reward of certification, and producers will not participate. Therefore, certification is not awarded for achieving ecological sustainability per se, but for demonstrable progress toward that goal. The hope is that, as companies make progress, and as certification gains profile and generates market demand, it will be possible to gradually raise the standards.

Another problem is that ENGO-led certification systems like FSC face competition from industry-led schemes that have arisen in their wake. The industry-led standards tend to be easier to achieve, but this is not obvious to average consumers who lack the technical knowledge needed to discern the differences (Schepers 2010). To counter this problem, several of the larger ENGOs work behind the scenes to promote FSC in the marketplace.

One of the main organizations promoting product certification is the World Wildlife Fund (WWF). Recognizing the overwhelming challenge of educating millions of individual customers, WWF has focused its efforts on developing partnerships with strategically selected companies. The aim is to help these companies "achieve positive and measurable benefits for their businesses, while creating conservation impacts where they matter most" (WWF 2018). Part of WWF's efforts include promoting the use of certified products, though it also works with companies on other aspects of their business.

Other groups, such as Greenpeace, also interact with businesses but tend to use a more coercive approach. A favoured tactic is to generate demand for certified products through selective targeting of high-profile retailers, which represent points of leverage. Consumers have a low tolerance for poor environmental performance and will readily punish retailers for such failings, even more than they will reward companies for positive performance (O'Rourke 2005). In addition, corporate brand and reputation are highly valued, so the involvement of even a relatively small percentage of customers can have substantial influence. Finally, the changes needed for achieving sustainability are less onerous for retailers than they are for resource companies, which means there is less resistance to change.

An illustrative example is the market action campaign against Victoria’s Secret by ForestEthics (now called Stand.earth) and its partners in the early 2000s. Targeting a company that sells sexy underwear may seem like an odd choice for a forest conservation initiative, but it was actually highly strategic. Victoria’s Secret is a business that sells not only underwear but an image and a lifestyle. In the 2000s, its well-known catalogue was a key component of its marketing strategy, reaching over 350 million customers each year (Merrick 2006). Producing a catalogue run of this magnitude involved killing a lot of trees—an image that did not square well with the company’s brand or its customer base, once the connection was made. These factors made the company an ideal target within a broader campaign to convert the entire catalogue industry to FSC-certified paper. After two years of campaigning, including store-front demonstrations and provocative ads (Fig. 3.10), Victoria’s Secret agreed to increase the recycled content of its paper and to give preference to FSC-certified sources. Three years later, an independent audit determined that 88% of paper purchased by Victoria’s Secret was either FSC-certified or post-consumer waste paper (Merrick 2006).

Fig. 3.10. This ad was used as part of ForestEthics’ “Victoria’s Dirty Secret” campaign. Credit: ForestEthics (now Stand.earth).

A campaign like the one against Victoria’s Secret has ripple effects that extend well beyond the company itself. Other companies, realizing that they may be next in line, often begin adjusting their practices proactively, resulting in the evolution of normative standards. The rise in demand for certified wood products, in turn, helps to create a business case for sustainable practices and certification among resource companies.

Promoting sustainability through market action is more challenging in the mining and oil and gas sectors because it is difficult to certify and trace these types of raw materials. Market-based campaigns are still utilized but in a more limited context. One approach has been to punish perceived laggards, again hoping for broader ripple effects. An example is the 2002–2004 market campaign by Greenpeace against ExxonMobil over its resistance to climate change mitigation efforts (Gueterbock 2004).

As with Victoria’s Secret, the choice of ExxonMobil was strategic. Not only was the company a perceived laggard, it had a large retail presence through its Esso gas stations that created a point of vulnerability (Gueterbock 2004). In this case, ExxonMobil did not capitulate, even though the campaign was supported by many motorists. This outcome is perhaps not surprising. The costs of sustainability are much higher for resource companies than they are for retailers like Victoria's Secret, which only had to switch suppliers. So there is more incentive to resist. Nevertheless, the campaign did damage ExxonMobil's brand, and affected sales to some (unknown) degree, sending a message to other companies that ignoring environmental concerns comes with tangible risks.

One further avenue of market influence on environmental protection is sustainable investing, also known as socially responsible investing (Glac 2009). Sustainable investing has been growing rapidly, and according to the US Sustainable Investment Forum, over $8 trillion is now invested in US funds that incorporate sustainability criteria (SIF 2022).

The scope of sustainable investing is broad and generally includes environmental, social, and corporate governance criteria. Companies are differentiated by these criteria and those with the best score within each industrial sector are rewarded with increased investment demand. In contrast to product certification systems, the assessment criteria for sustainability are not standardized and no attempt is made to determine whether a company's practices are in fact sustainable. Companies are simply rewarded for doing better than their peers, and this provides the business case for improving environmental practices (Glac 2009).

Little information currently exists as to how effective sustainable investing has been in terms of actually changing company practices. Moreover, over the past few years there has been a growing backlash against the perceived misrepresentation of investment products by fund managers who stand accused of simply washing everything in green (SIF 2022). As a result, regulators and sustainability advocates are now placing increasing pressure on fund mangers to improve transparency and accountability with respect to the sale of sustainable investing products. Consider it a work in progress.

Variability in Company Attitudes and Approaches

Resource companies vary widely in their commitment to sustainability and the conservation of biodiversity. Some are progressive and willing to undertake meaningful actions, whereas others are quite recalcitrant, refusing to do more than is absolutely required.

An important determinant of environmental attitudes is company size (Hillary 2004). Small resource companies tend not to attract much public attention, largely because ENGOs usually focus their limited capacity on large companies, which make a more efficient target (Hillary 2004). Individually, large companies have more environmental impact than small companies, and they are more vulnerable to market action. Public tolerance for the environmental failings of large multinational corporations tends to be quite low, whereas the transgressions of small locally owned companies may be forgiven. Finally, small companies have less technical capacity and flexibility than large companies for implementing new approaches. Consequently, leadership in environmental practices is most often found among large resource companies (which is not to say that all large companies are progressive). Conversely, small resource companies generally prefer doing things the old way as long as possible, though there are exceptions.

Environmental attitudes are also affected by company-specific factors. For example, a forestry company with ample wood supply is more likely to embrace progressive practices than a company facing a shortfall. Senior management is another important factor. These individuals make the key decisions that set a company's direction, so individual personalities can make a big difference. Decisions are influenced by the experience, risk tolerance, and personal biases of those involved. Thus, two companies faced with similar circumstances may come up with quite different conclusions about the best course of action.

Company attitudes and approaches toward conservation are also shaped by industry associations. Associations can have a positive effect by articulating industry norms and facilitating the dissemination of new ideas to their member companies. However, externally, industry associations can be a regressive force. The positions they present as the voice of industry in public policy debates often gravitate to the lowest common denominator among the companies they represent.

Significant differences also exist among industrial sectors. These will be discussed in Chapter 7 in association with our examination of sector-specific approaches to biodiversity conservation.

Indigenous Communities

Indigenous influence on conservation is mediated mainly through Indigenous involvement in land and resource planning. Indigenous people have treaty rights that must be respected, and the courts have clarified that governments and resource companies have a legal duty to consult with Indigenous communities about proposed developments (SCC 2017a). The duty to consult does not provide a veto over developments, but it does give Indigenous communities considerable influence over what happens. In northern areas, land claims agreements have provided additional rights that have led to various forms of collaborative planning (Wyatt et al. 2013).

Contemporary Indigenous conservation attitudes and practices are the legacies of adaptive strategies that arose through cultural evolution (Berkes et al. 2000; Smith and Wishnie 2000). These practices were developed at a time when nature was still firmly in control, limiting the size of human populations to levels the local environment could sustainably support (Johannes 2002; Diamond 2005).

The most common form of cultural evolution occurs through incremental learning or "fine tuning" (Turner and Berkes 2006). Like genetic mutations under natural selection, new ideas and practices arise spontaneously and may become adopted as cultural traditions if they improve the prosperity of the community (Berkes et al. 2000). Successful practices need not involve conservation, but in subsistence economies this will often be the case because of the strong dependence of communities on their local environment (Gadgil et al. 1993).

Conservation-related practices are most often associated with resources that are predictable, amenable to control, and important to the community (Winterhalder and Lu 1997; Smith and Wishnie 2000; Berkes 2012). These conditions are often met with plant resources, and this is where some of the clearest examples of Indigenous conservation practices are found. An illustrative example is the removal of bark from cedar trees for making clothing, mats and baskets. Traditional practice demands that harvesters exercise restraint and only remove one or two straps from each tree, so as to keep the tree alive (Turner and Berkes 2006).

Another form of cultural evolution involves so-called "crisis learning," which typically arises under novel or changing conditions, such as migration into a new area or the development of new technology (Johannes 2002). Sometimes crises promote rapid adaptation, leading to sustainable practices that become ingrained in tradition. In other cases, animals and plants are extirpated, either because the rate of cultural adaptation is too slow or because the affected species are not important to the community (Johannes 2002).

Crisis learning has been most clearly documented in island societies; however, the selective extinction of North American megafauna after initial human migration to the continent suggests that it occurred here as well (Surovell and Grund 2012). There are also cases where human societies collapsed alongside their depleted resources. The demise of the Easter Island people serves as the poster child for this outcome but is not an isolated example (Diamond 2005).

The association of crisis learning with changing conditions highlights an important characteristic of cultural evolution and adaptive processes in general. Strategies that are highly effective under one set of conditions offer no guarantee of success under novel conditions (Diamond 2005). The implication is that traditional Indigenous practices are not guaranteed to remain effective in the modern world, which has changed tremendously since the pre-contact era.

A major challenge relates to the transition from a subsistence lifestyle to participation in a market economy. Just like the rest of Canadian society, Indigenous communities today are torn between achieving their environmental objectives and achieving their economic objectives. Finding the right balance is not easy, and there have been divisions within and among communities (Conklin and Graham 1995). For example, Indigenous communities lined up on both sides of the debate over the construction of the Trans Mountain pipeline in BC (Tasker 2017).

The growth of Indigenous populations and the advent of advanced technology are two additional features of the modern world that the traditional Indigenous approach to conservation is ill-equipped to handle (Fig. 3.11). Caribou hunting practices provide a case in point. Indigenous people have been hunting barren-ground caribou in northern Canada for millennia, with checks and balances in place to accommodate natural fluctuations in the size of the caribou herds. Whenever herds became scarce, they were harder to find, forcing Indigenous communities to rely on alternative sources of meat (Nesbitt and Adamczewski 2013). This provided caribou with time to recover.

Fig. 3.11. An Inuit hunter, illustrating the modern equipment that Indigenous hunters commonly use today. Credit: A. Walk.

Today, hunting practices have changed, and the old checks and balances do not always work. Hunting methods are highly effective and, using fast snowmobiles, trucks, and airplanes, hunters can find caribou herds even when numbers are very low (Boulanger et al. 2011). Indigenous leaders in the NWT have also observed that "some hunters, particularly younger hunters, have at times used practices less respectful of caribou," including killing more than is needed and not using all of the animal (Nesbitt and Adamczewski 2013, p. 18). These changes have led to unsustainable rates of harvest and have been a major contributor to the precipitous declines of some herds (Boulanger et al. 2011). For example, the Bathurst herd in the NWT, which numbered over 400,000 individuals in the 1980s, has been reduced to just 20,000 individuals in 2015—a modern example of crisis learning (GONWT 2016).

A contrasting example from the same region is the 2006 Dehcho land-use plan, which illustrates the positive aspects of Indigenous conservation. The Dehcho First Nations produced the plan to promote the social, cultural, and economic well being of residents and communities in the 210,000 km^2 Dehcho territory, located in the NWT adjacent to the border with BC and Alberta (DLPC, 2006).

What is notable about the Dehcho plan is that, although it is designed to achieve multiple societal objectives, including economic development, 50% of the land is zoned for conservation where only tourism and traditional Indigenous use are permitted. The plan is also notable for its blending of traditional views with modern science-based concepts such as cumulative effects management, the natural range of variation, and the precautionary principle (see Chapter 7). Unfortunately, the plan remains mired in negotiations over land use between the Dehcho and the federal and territorial governments.

In conclusion, contemporary Indigenous approaches to conservation and land management demand a balanced view. The twentieth-century portrayal of Indigenous people as masters of ecological sustainability (see Chapter 2) is an oversimplification. Traditional conservation practices were developed under a subsistence lifestyle and are not always applicable to modern circumstances. Moreover, Indigenous communities today must grapple with trade-offs between environmental objectives and economic objectives, and this can influence their decisions regarding land use. But on the whole, Indigenous communities generally remain strong proponents of conservation, with a close connection to the land and respectful attitudes toward nature.

It is also important to recognize that the Indigenous concept of conservation is distinct from the modern concept of biodiversity conservation as described in this text. In particular, Indigenous conservation is more focused on sustainable use than on the protection of species at risk (Smith and Wishnie 2000). Notably, Indigenous communities in Nunavut have been systematically blocking the official listing of species at risk that reside in the territory, denying them the protections afforded by the *Species at Risk Act* (Findlay et al. 2009). Finding common ground, in the context of conservation planning, can be accomplished, but is often challenging.

Government

In an idealized view, the government serves as a referee that ensures a balance is achieved among competing public interests. It does this by providing a forum for dialog among parties and by facilitating learning about potential solutions for resolving conflict. When conflict is unavoidable, the government takes control, making executive decisions that direct and constrain the actions of the parties involved in ways that best serve the broad public interest.

In practice, governments are not neutral arbitrators of the public interest (Wood et al. 2010). Political parties develop policy platforms that define where they stand on various value-laden issues and then place these platforms before the electorate. Individual conservation issues rarely have enough political profile to merit inclusion in such platforms, but general positions that impinge on conservation, such as being "pro-business" or "pro-environment," are fairly common. In addition, governments have a responsibility for managing the economy and providing services. This means they have a direct stake in many resource management decisions. Finally, governments face many challenges that limit their ability to make effective decisions, as described by Wilson (1998):

> Governments the world over muddle through. They try to plan, but mostly they react. They spend a fair bit of time grappling with states of full or partial paralysis brought on by uncertainty, inadequate information and capacity, internal divisions, and conflicting advice and pressures. They are frequently forced to wrestle with circumstances beyond their control, or with the unintended and unhappy consequences of decisions they have made. For the most part, they move incrementally. … Overwhelmed by the complexity of the problems they confront, decision makers lean heavily on pre-existing policy frameworks, adjusting only at the margins to deal with the distinctive features of new situations. Occasionally, when the planets are aligned, governments seize the opportunity to consolidate disparate policy tendencies into a coherent shift in policy direction. (pp. 334–335)

Because most conservation decision making occurs within the public sphere, our system of governance, messy as it is, is of central importance to the practice of conservation. Conservationists need to understand how the system operates and how decisions are made in order to be effective (Clark 2001).

It is useful to differentiate three basic forms of government decision making. Real-world decisions do not always fall neatly into these three categories, but understanding these basic forms provides a foundation for understanding the full range of complexity that exists:

1. **Routine management.** This form of decision making applies to routine operational decisions. Here, the scope of decision making is tightly circumscribed by existing laws, policies, and plans. The decision makers are typically managers working alone or in small groups. An example is the setting of annual harvest limits for game species.

2. **Planning.** Management issues that are not routine—especially problems in need of a solution—require a more complex form of decision making. Decisions of this type often take the form of plans and strategies and are best made using a structured decision-making framework (see Chapter 10). Stakeholders are often involved in this type of decision making, and there may or may not be some form of input from the general public. Governments retain

ultimate control over decision making, sometimes at the bureaucratic level and sometimes at the ministerial or cabinet level. An example is the development of species recovery plans.

3. Policy development. The highest level of decision making is associated with the development of new policies and legislation. In contrast to planning, policy development rarely follows a well-defined, structured process. Policy dialog occurs in the public sphere and includes the media, a wide range of stakeholders, and the general public. Governments ostensibly control the process, but in practice may struggle to maintain autonomous action. The locus of government decision making generally resides with cabinet. An example is the development of the *Species at Risk Act*.

In the following sections, we will examine the policy development process in detail and then briefly review the current conservation policy landscape. We will defer discussion of routine management and planning to Chapter 10, where we will examine the mechanics of structured decision making in detail.

Policy Equilibrium

Studies of policy development reveal a nonlinear pattern of change (Baumgartner et al. 2014). Most of the time, policies are essentially locked in place, subject to only slow incremental adjustments. These long periods of stasis are interspersed with sporadic bursts of more rapid change. This unpredictable pattern is a hallmark of complex systems, in which interactions among many agents result in both positive and negative feedback processes and complex dynamics (Klijn 2008).

During the periods of policy stasis, the system is not quiescent. Like an ecological system in equilibrium, individual actors are highly active, yet the overall system remains relatively unchanged because of the stabilizing influence of negative feedback processes. We will examine these stabilizing processes first, then turn our attention to the causes and consequences of instability and rapid policy shifts.

When a policy system is in a stable state, issues tend to be captured within subsystems composed of small groups of bureaucratic specialists and key stakeholders working largely outside of the public eye (Lee 1993). The existence of these decentralized policy subsystems provides the overall system tremendous parallel processing capacity, making it possible for governments to respond to the myriad issues confronting it. These groups maintain attention on problem areas, even when the public, media, and elected officials are focused elsewhere. Moreover, they have the time and technical capacity needed for substantive learning and problem solving to take place (Lee 1993). These groups are a key entry point for conservation practitioners to contribute to the policy process.

A characteristic feature of policy subsystems is that substantive shifts in policy are difficult to achieve. Solutions to problems tend to be sought within the existing policy framework or through minor modifications of it. Processes that promote stability predominate and constitute barriers to change.

One reason the status quo tends to be maintained is that existing policies often represent compromise solutions to intractable conflicts. In such situations, policies reflect the balance of power that exists between the opposing sides of an issue. So long as this balance of power remains in place, substantive changes in policy are unlikely to occur. Consider the *Species at Risk Act*—a compromise solution to species protection that took years of acrimonious debate to achieve. Any attempt by environmentalists to strengthen the Act, or by industry to weaken it, would quickly result in the mobilization of opposition.

Another important factor promoting the status quo is the legacy of earlier decisions. Decisions made in the past often constrain what is possible in the present. This is referred to as **path dependency** (Ingram and Fraser 2006). For example, once a forest is allocated, a mill is built, and a local community is established, it is all but impossible to roll back the clock. Because of these legacies, strategies that would otherwise be ideal for achieving contemporary management objectives are often infeasible.

There are also several institutional factors that contribute to policy inertia (Lee 1993; Clark 2002):

- Policy subsystems are largely populated by career bureaucrats who may find substantive change threatening because it challenges their worldview and values or because it has the potential to upend existing power and control structures.
- Faced with changing conditions, bureaucrats may remain wedded to existing approaches if viable alternatives are not available or simply because they fear making mistakes in uncharted territory. Government reward structures provide little incentive for risk taking.
- Policy subsystems cannot make significant changes to policy without the support of political leaders in cabinet. But cabinet can only focus on a small number of priority issues at any given time because of capacity constraints. This limits the overall pace of change in the system.
- Governments tend to be most responsive to conflict, which means that "slow-creep" issues, such as habitat deterioration, tend to receive little attention until a crisis point is reached. This problem is exacerbated by the short planning horizons imposed by electoral cycles.
- The development of new policy requires staff and financial resources that are usually difficult to obtain. Over the years, federal and provincial funding of resource management has been in decline.
- Policy subsystems tend to monopolize specific areas of policy, and communication and coordination among these policy "silos" are often limited. Changes in policy direction within one subsystem may be perceived by others as an invasion of turf, and resisted. This is exacerbated by the fact that many policy subsystems work at cross-purposes with each other because of conflicting objectives (e.g., environmental protection vs. economic development).

There are also political barriers that impede the advancement of conservation (Wood et al. 2010). Governments, particularly at the provincial level, have a strong incentive to promote industrial development because the associated rents and taxes are needed to balance their budgets. Also, many politicians perceive employment levels and economic health as decisive election issues, overshadowing environmental issues (VanNijnatten 1999). The tendency to favour development can be exacerbated when governments are elected on the basis of narrowly-defined wedge issues and make little effort to govern in the broad public interest (Wood et al. 2010). The public interest concerning the environment can also be overridden as a result of industry influence, which occurs through campaign funding, intensive lobbying, and the interchange of personnel (Meghani and Kuzma 2011).

Policy Change

The periods of policy equilibrium are not entirely static. Public values change, the balance of power shifts among policy actors, new information and ideas come to light, policy subsystem participants turnover, and the consequences of unresolved problems accumulate. Policy subsystems deal with these changes mainly through incremental adjustments to existing policy. Like compound interest, these small adjustments accumulate over time. When viewed over long enough periods (decades), most policies can be seen to undergo a transformation, albeit in slow-motion.

Policies can also, on occasion, undergo episodes of rapid change. These episodes of relative instability result from positive feedback processes that are intrinsic to the system but manifest only under certain conditions (Baumgartner et al. 2014). For example, recall from Chapter 2 how the burning Cuyahoga River attracted national media attention in 1969, whereas earlier episodes of burning were completely ignored.

The basic mechanics of these positive feedback processes are well known; however, they are subject to complex interactions that make it all but impossible to predict specific outcomes (Klijn 2008). This means that policy breakthroughs cannot be engineered. The most that proponents of specific objectives can do is to push the system in the right direction and hope for the best.

For a given policy to undergo rapid transformation, it must first escape the gravitational pull of the policy subsystem it is normally bound to (Ingram and Fraser 2006; Baumgartner et al. 2014). Once on the macro political stage, other policy actors may engage, and major shifts in policy direction are possible (though not guaranteed). The emergence of a high-profile political champion can make a big difference. The question is: how and why does this shift to the macro political stage take place?

In most cases, political stalemates are broken by new ideas and by changes in power structures that lead to increased support for alternative perspectives (Ingram and Fraser 2006; Baumgartner et al. 2014). Ideas are important because they form the basis of policy images or frames—the shorthand narratives that define how we think about individual issues. For example, wolves as a menace vs. wolves as a symbol of wilderness. When existing frames are challenged by shifting values and new ways of thinking, policy upheaval becomes possible. Defenders of the status quo may find themselves holding solutions to the wrong problem. In addition, factual information may accumulate that challenges claims about the effectiveness of existing approaches. Finally, new and superior solutions to management issues may be developed through learning and debate during periods of policy stasis. In this case, defenders of the status quo may find themselves wielding ideas and concepts that are discredited or outmoded.

Although new ideas may challenge the status quo and weaken its intellectual foundation, this in itself does not guarantee that change will occur. Policy alternatives also need visibility and an expanding base of support if they are to attract attention on the macro political stage. Sometimes this is achieved through the development of new alliances among policy actors, as occurred during the War in the Woods (see Chapter 2). But what really tends to attract political attention is public engagement, and this usually requires involvement of the mass media (Fig. 3.12).

Fig. 3.12. Mass media is a major conduit for information about environmental issues.

The media not only disseminates information to the masses, it also filters and processes the information it provides. As a result, the media plays a critical role in setting political agendas. As noted by Soroka (2002, p. 265), "The press may not be successful much of the time in telling people what to think, but it is stunningly successful in telling readers what to think about." This is particularly true with environmental issues, which most people learn about through the media, rather than through direct experience.

Over the years, the media has served as the main catalyst for amplifying public concerns about environmental issues (Hansen 2010). However, the media should not be thought of as an environmental advocate. It simply reports on what it considers to be news, and conservationists quickly learn this rarely includes the issues they hope to draw attention to. There is a bias toward conflict, novelty, and topics that are currently in vogue.

Because of its critical role in agenda setting, the media is frequently the target of manipulation efforts. Environmentalists, in particular, have become adept at "playing the game," using manufactured conflict (e.g., demonstrations, blockades, etc.) and other measures to draw attention to issues that would otherwise moulder in obscurity. Industry cannot respond in kind, but aided by public relations firms, it is skilled at reframing controversies in a more benign light. It cherry-picks success stories and uses feel-good advertisements to send the message that environmentalists are misinformed or prone to overreaction. As a result, public dialog about environmental issues is often reduced to political theatre involving simplified caricatures of what is really at stake. This being the case, government decision makers are often more interested in the amount of news coverage an issue gets than precisely what is being said (Mazur 1998).

An important feature of media coverage is the potential for positive feedback. A virtuous cycle can ensue where mass media coverage of an issue generates broad public concern, which in turn increases the likelihood of more media attention (Soroka 2002). Politicians, sensing a sea change, may then respond, further legitimizing the issue and raising its profile even higher. For issue proponents, this is the Holy Grail. The problem is that it is difficult to engineer; public interest cannot be forced. The issue must have cultural resonance, linking to people's interests, concerns, and fears, all of which change over time (Hansen 2010). Moreover, issues interact with each other. Sometimes circumstances may propel an issue to the forefront, but more often issues must struggle for visibility among other competing themes. So timing is a critical and unpredictable factor.

In recent years, the media landscape has been changing, with the advent of social media and declining viewership of traditional news sources. The effects this will have on the development of conservation policy are not yet fully understood. On the one hand, conservationists are much less reliant on the mainstream media for disseminating their messages. On the other hand, the increase in media options has fragmented viewership, making it harder to reach a mass audience. Moreover, media fragmentation seems to have fostered societal "echo chambers," and this hinders the development of a common understanding of issues, as required for informed public debate (Williams et al. 2015).

Arrival on the macro political stage is a prerequisite for substantive policy change; however, change is far from assured. Federal and provincial cabinets cannot deal with all of the demands they are confronted with, so issues must compete for attention (Baumgartner et al. 2014). Conflict, crisis, and high public profile tend to boost salience. Party ideology and campaign platforms also influence the prioritization of issues. Finally, political leaders view issues more holistically than issue proponents. Evidence of policy failure is itself usually insufficient to cause policy change (Ingram and Fraser 2006). Policy solutions must be available, and there must be a reasonable prospect of success. Assessments of political feasibility take into account the strength of opposing forces, the amount of change likely to be tolerated, and potential gains and losses in political capital. Consideration is also given to how potential changes in policy are likely to impact the rest of the system, especially in economic terms.

Ultimate outcomes are difficult to predict. Prominent issues that are not aligned with government interests and priorities may be ignored, delayed, or denied. Token offerings are sometimes made, containing only the appearance of change. However, governments do not have absolute control over the process. At this level, policymaking "is mostly about expediting or delaying the way that immutable forces unfold, or about nudging the resultant trajectories a few degrees to one side or another" (Wilson 1998, p. 344). The point is, important issues and ideas can rarely be repressed indefinitely. Political pressures build and must ultimately find release. The government's hand may eventually be forced, or more likely, opposition parties may seize the issue as a political opportunity and move it forward once in power. For their part, proponents of the status quo will do whatever they can to prevent or slow change. Issues without sustained public interest and support may end up back within rigid policy subsystems.

The evolution of modern conservation policy has been typical of the processes described. Since it emerged as an issue in the 1970s, biodiversity conservation has been and continues to be a challenger of the status quo, resisted by various forces opposed to change. There have been periods of rapid change, in the early 1970s and early 1990s, but most of the time conservation policy has evolved through incremental adjustments within policy subsystems. We are currently in a period of policy stasis, reflecting a political stalemate between proponents of conservation and proponents of economic development. Notably, climate change has emerged as the dominant environmental issue of our time, leaving much less space on the macro political stage for other environmental concerns.

Current Conservation Policy

The *Canadian Biodiversity Strategy*—Canada's response to the UN *Convention on Biological Diversity*—continues to serve as the foundation for conservation policy in Canada. The Strategy defines conservation as "the maintenance or sustainable use of the earth's resources in order to maintain ecosystem, species and genetic diversity and the evolutionary and other processes that shape them" (EC 1995, p. 6). This is a biocentric rather than a utilitarian interpretation of conservation.

Since the release of the Strategy in 1995, an intergovernmental **Biodiversity Working Group**, chaired by Environment and Climate Change Canada, has worked to identify priorities, engage with Canadians about implementation, and report on progress. In 2015, the Biodiversity Working Group released the *2020 Biodiversity Goals & Targets for Canada* (see Box 3.1, below), endorsed by all federal, provincial, and territorial governments (BWG 2015). This document emphasizes ecosystem approaches to conservation, including sustainable industrial practices and additional protected areas. Species at risk are included in the context of recovery planning but genetic diversity is not specifically mentioned.

As of 2023, the *2020 Biodiversity Goals & Targets for Canada* statement continues to serve as the federal policy on biodiversity conservation. However, the target for habitat protection has increased to 30% of Canada's lands and oceans by 2030.

To be clear, the goals and targets that have been articulated do not represent government commitments; they are statements of what we would like to accomplish. The document uses the term "aspirational" to describe them. Furthermore, most of the targets are actually strategies and not measurable outcomes. Policy statements like this provide direction, but they do not compel action or define accountability in the way that legislation does (AGC 2013). This is not to say that policy direction is unimportant. Government programs and budget allocations are generally linked to established policies. Policies also guide planning and routine management by defining the objectives that are to be accomplished.

Though all the provinces were involved in the development of the 2020 goals and targets, their support has been quite variable. Some provinces, notably Ontario and BC, have demonstrated a serious commitment to conservation through complementary biodiversity strategies of their own (BCMOE 2009; OMNR 2012). Uptake in most of the other provinces has been inconsistent. Support sometimes exists only within the ministry responsible for wildlife management. Without broad-based support within government and meaningful efforts at policy integration, conservation measures are difficult to implement. Consequently, despite national commitments, a large gulf remains between conservation intent and conservation practice, with substantial variability across the country.

In addition to formal biodiversity strategies, conservation policy exists under the rubric of forest management, and to a lesser extent, water management. As we saw in Chapter 2, following the War in the Woods, most provinces transitioned to a sustainable forest management paradigm that includes the maintenance of forest biodiversity as a central outcome. National leadership has been provided by the Canadian Council of Forest Ministers, which has developed a set of criteria and indicators of sustainability (see Chapter 7). Adoption and implementation of these criteria and indicators have again been quite variable among provinces and from company to company.

Another area of conservation policy applies to the management of species at risk. In jurisdictions where species at risk legislation exists, policy is used to provide the detail needed for implementation. In provinces lacking such legislation, or where legislation is rudimentary, policy provides the primary basis for managing species of risk.

Current Conservation Legislation

In contrast to conservation policy, which emphasizes ecosystem approaches, conservation legislation in Canada is focused on species. Canada's flagship conservation law is the *Species at Risk Act* (SARA), passed by Parliament in 2002 (See Chapter 2). The purpose of SARA is to "prevent wildlife species from being extirpated or becoming extinct, to provide for the recovery of wildlife species that are extirpated, endangered or threatened as a result of human activity, and to manage species of special concern to prevent them from becoming endangered or threatened" (GOC 2002, Sec. 6). SARA is one of the few federal laws (or Canadian environmental laws generally) that imposes strong, court-enforceable duties on government to protect wildlife and its habitat. Nevertheless, it is still a compromise solution subject to a variety of limitations. We will have a closer look at the strengths and weaknesses of SARA in Chapter 6.

Provincial commitments to the protection of endangered species are again quite variable. Seven provinces and territories have dedicated species at risk legislation, whereas others manage species at risk under older wildlife legislation (SPI 2018). Ontario's *Endangered Species Act* is the strongest and includes most of the strengths and weaknesses of SARA (Ecojustice 2012a). In other provinces, species at risk legislation is generally weaker than SARA, featuring greater political discretion and fewer firm commitments (SPI 2018). Adding to the general confusion over jurisdictional roles, the species at risk identified at the provincial level are not necessarily the same as those identified at the federal level. The provinces are also further behind than the federal government in terms of implementation.

No legislation comparable to SARA exists for conservation at the ecosystem scale. Instead, various aspects of biodiversity conservation have been incorporated, to a greater or lesser degree, into laws that serve a broader purpose. The most prominent example is the incorporation of biodiversity objectives into legislation governing forestry, which has occurred in some provinces but not all. For example, Ontario's *Crown Forest Sustainability Act* (GOO 1994, Sec. 2.3) includes the following principle: "large, healthy, diverse and productive Crown forests and their associated ecological processes and biological diversity should be conserved." The biodiversity component of such forestry laws tends to function much like high-level policy. The language used establishes intent and direction but is vague enough to preclude successful court challenge.

Other legislation that helps support conservation includes (1) laws governing land-use planning, water management, and sustainable development; (2) laws concerning environmental assessments; and (3) provincial laws related to wildlife and game management (Boyd 2003). Also important are laws supporting the establishment and management of protected areas, which come in many different forms (see Chapter 8). The federal *National Parks Act* is the most rigorous in Canada in terms of supporting conservation and includes explicit requirements for maintaining ecological integrity.

In conclusion, Canada's conservation laws and policies do much to clarify intent but provide few definitive commitments. They represent society's collective conservation aspirations, and this is crucial for guiding what we do as conservationists. However, they do not reflect the practical realities of conservation. As we will see in the case studies (Chapter 11), the true meaning of conservation is revealed when conflict occurs, requiring trade-off decisions to be made (McShane et al. 2011). At this point, policy commitments often fade into the background and the practical exigencies of local circumstances rise to the fore. This is something that conservation practitioners must be prepared for.

Box 3.1. Summary of the 2020 Biodiversity Goals and Targets for Canada

Goal A. By 2020, Canada's lands and waters are planned and managed using an ecosystem approach to support biodiversity conservation outcomes at local, regional and national scales.

> **Target 1.** At least 17 percent of terrestrial areas and inland water, and 10 percent of coastal and marine areas, are conserved through networks of protected areas and other effective area-based conservation measures.
>
> **Target 2.** Species that are secure remain secure, and populations of species at risk listed under federal law exhibit trends that are consistent with recovery strategies and management plans.
>
> **Target 3.** Canada's wetlands are conserved or enhanced to sustain their ecosystem services through retention, restoration and management activities.
>
> **Target 4.** Biodiversity considerations are integrated into municipal planning.
>
> **Target 5.** The ability of Canadian ecological systems to adapt to climate change is better understood, and priority adaptation measures are underway.

Goal B. By 2020, direct and indirect pressures as well as cumulative effects on biodiversity are reduced, and production and consumption of Canada's biological resources are more sustainable.

> **Target 6.** Continued progress is made on the sustainable management of Canada's forests.
>
> **Target 7.** Agricultural working landscapes provide a stable or improved level of biodiversity and habitat capacity.
>
> **Target 8.** All aquaculture in Canada is managed under a science-based regime that promotes the sustainable use of aquatic resources in ways that conserve biodiversity.
>
> **Target 9.** All fish and invertebrate stocks and aquatic plants are managed and harvested sustainably, legally and applying ecosystem-based approaches.
>
> **Target 10.** Pollution levels in Canadian waters, including pollution from excess nutrients, are reduced or maintained at levels that support healthy aquatic ecosystems.

Target 11. Pathways of invasive alien species introductions are identified, and risk-based intervention or management plans are in place for priority pathways and species.

Target 12. Customary use by Indigenous peoples of biological resources is maintained, compatible with their conservation and sustainable use.

Target 13. Innovative mechanisms for fostering the conservation and sustainable use of biodiversity are developed and applied.

Goal C. By 2020, Canadians have adequate and relevant information about biodiversity and ecosystem services to support conservation planning and decision-making.

Goal D. By 2020, Canadians are informed about the value of nature and more actively engaged in its stewardship.

CHAPTER IV

THE SCIENTIFIC DIMENSION OF CONSERVATION

The Scientific Dimension of Conservation

The Evolution of Conservation Science

The take-home message of the last chapter is that conservation is a conflict-driven social and political process. But there is more to the story than just the interplay of values and political forces. This is a field where ideas and knowledge matter, influencing both the "what" and "how" of conservation (Wilson 1998). With this in mind, we now turn our attention to the role of science in conservation. We will begin by reviewing the evolution of conservation science and then consider how science is applied in practice, and by whom. Along the way, we will explore two controversial topics: the advent of "new conservation," linked to the concept of ecosystem services, and the fine line separating conservation science and conservation advocacy.

The field of conservation science traces its origin to scientists responding to concerns about declining game species and forests in the early twentieth century (MacDowell 2012). The application of biological science to these natural resource problems led to the emergence of fish and wildlife management and forestry as distinct applied science disciplines.

As we saw in Chapter 2, public attitudes toward nature underwent a profound transformation in the 1960s and 1970s. Non-consumptive values rose to the forefront, and conservation began to mean the maintenance of biodiversity rather than the sustainable harvest of game species and forests. This shift in public values led to a schism within the scientific community, with some researchers continuing to orient toward utilitarian management objectives and others redirecting their efforts toward broader biodiversity goals. The breakaway group eventually formed its own body—the Society for Conservation Biology—and by the mid-1980s, conservation biology was established as a distinct subdiscipline of biology (Meine et al. 2006).

The objectives of conservation biology were formalized as a mission statement by the Society for Conservation Biology at its inaugural meeting in 1986: “to help develop the scientific and technical means for the protection, maintenance, and restoration of life on this planet—its species, its ecological and evolutionary processes, and its particular and total environment” (Meine et al. 2006, p. 637). This mission statement directed the field’s research agenda and provided the context for making management recommendations (Robinson 2006). Thus, from the outset, conservation biology was a **normative** discipline—it embraced certain values and sought to apply scientific methods to advance those values.

Early conservation biologists incorporated the conservation approaches developed by the resource management community and took the field in new directions. The initial emphasis was on species extinction, motivated in large part by demands of the US *Endangered Species Act*. Prominent research topics included (Soule 1986):

- Population viability analysis and minimum viable population size
- Conservation genetics and captive breeding
- Habitat fragmentation and habitat restoration
- Island biogeography and protected area design
- Keystone species and community stability

These topics remained central to conservation biology as it matured, but the field also expanded into new areas. From its initial focus on populations, research began to address the conservation of biodiversity more broadly, especially at the landscape and regional scales. A new understanding of what conservation entailed and what was required of science also emerged. At the outset, many had assumed that policy commitments to maintain biodiversity could be taken at face value, implying that conservation was mainly a matter of “figuring out the biology.” In time, it became clear that conservation was fundamentally a social process, and that trade-offs with other land-use objectives had to be considered when crafting conservation solutions, policy commitments notwithstanding (Kareiva and Marvier 2012).

Conservation biology responded to these challenges by widening its tent, becoming ever more interdisciplinary. The initial core of researchers, whose expertise was mainly in biology and resource management, was augmented with researchers from a range of social sciences, including environmental economics, political science, environmental law, and environmental ethics. There was also increasing effort to synthesize the growing body of conservation literature into principles and application frameworks.

Ecosystem Services and “New Conservation”

The early 2000s saw the emergence of the ecosystem services concept, referred to by some proponents as “new conservation” (Daily et al. 2009). The intent was to boost the success of conservation initiatives by addressing some of the perceived shortcomings of existing approaches. Proponents argued that the benefits of conservation needed to be better quantified so they could be integrated more effectively into decision making. Also, the scope of conservation needed to be broadened to include utilitarian benefits, rather than focusing mainly on the intrinsic and intangible values of biodiversity (Turner and Daily 2008; Kareiva and Marvier 2012; Marvier 2014). Only through such efforts could the generally poor track record of earlier conservation initiatives be improved.

The National Roundtable on the Environment and Economy (2003) articulated the concept in a Canadian context:

> We value nature for many reasons: not only does it have aesthetic and spiritual aspects, but it also provides us with clean air and water and other ecological services on which our economy, environment and quality of life depend. These ecological services are increasingly being seen as a natural form of capital that has economic value. … That much of our natural capital—from water to trees to oil and gas deposits—is available to the public and to industry at little or no cost has led to a perception that conservation is bad for jobs and bad for the economy. … not understanding these costs and benefits is compromising our ability to make meaningful decisions about the balance between nature conservation and industrial development. (pp. xiii, 40)

The ecosystem services concept quickly gained adherents, especially after it was profiled in the 2005 *Millennium Ecosystem Assessment*, commissioned by the UN (Reid et al. 2005). Methodologies were developed for quantifying the benefits of ecological services, and application frameworks were established (Turner and Daily 2008; Daily et al. 2009). An important feature of these efforts was that ecological benefits were all expressed in monetary terms, reflecting the economic foundations of the ecosystem services concept. Having all values expressed in the same units—dollars—facilitated cost-benefit analyses.

Box 4.1. An Ecosystem Services Success Story

An often-cited example of the potential of the ecosystem services concept involves the management of New York City's water supply (Daily and Ellison 2002). In 1989, the city was faced with an order from the Environmental Protection Agency to build a water filtration plant for the city, with an estimated price tag of \$6–8 billion. Filtration was needed because the quality of the surface water supplying 90% of the city's needs was declining as a result of population growth and development in the surrounding watershed. Instead of building the filtration plant, planners directed \$1.5 billion to a watershed conservation project. The project was a success, both in terms of maintaining water quality and avoiding the cost of filtration. As a bonus, it also helped to maintain the overall ecological integrity of the region.

The New York City watershed project was successful because there was a direct linkage between an important ecological service (water filtration) and the needs of a nearby population centre. The costs and benefits of the management options were also readily quantifiable. Applying the ecosystem services concept in more remote regions is much more challenging because the potential markets are distant and often just hypothetical. Simply extrapolating outcomes from one region to another is unlikely to provide economically meaningful results.

Following the rapid rise of the ecosystem services approach, and its adoption by many policymakers, opposition began to mount (Redford and Adams 2009; Doak et al. 2013). Many conservationists felt that the concept was being perceived as a panacea and would lead to unintended and undesirable consequences (Ostrom et al. 2007;

McShane et al. 2011). There was also alarm over the “new conservation” label being used by some proponents, implying that the ecosystem services approach supplanted existing conservation approaches (McCauley 2006; Soule 2013; Miller et al. 2014).

The fundamental problem, for those opposed to the concept, was that only utilitarian values could be meaningfully expressed in monetary terms (Spangenberg and Settele 2010). Attempts to put a dollar value on the intrinsic value of nature were not compelling. They relied on unrealistic assumptions and did not provide consistent or believable results (Nunes and van den Bergh 2001; Spangenberg and Settele 2010; Chan et al. 2012).

Given the impracticability of assigning a meaningful monetary value to the intrinsic and intangible values of biodiversity, many conservationists worried that the ecosystem services approach would marginalize these values (Redford and Adams 2009; Doak et al. 2013). Indeed, the proposed frameworks clearly emphasized utilitarian outcomes such as recreation, the maintenance of water quality, carbon sequestration, pollination, and erosion control (Turner and Daily 2008; Troy and Bagstad 2009; SC 2013). Opponents saw the concept as a slippery slope ending in the commodification of nature (Gomez-Baggethun and Ruiz-Perez 2011; Turnhout et al. 2013). As we saw in Chapter 2, treating nature as a commodity has never worked out well for biodiversity.

A fundamental problem is that most commonly cited ecosystem services do not depend on a natural ecosystem. These services are so generic that they can be supplied without native species or natural ecological structures. For example, a tree plantation can supply most ecological services (e.g., erosion control, carbon storage, production of oxygen) just as well as a natural forest (Fig. 4.1; Redford and Adams 2009). What it cannot do is support the full complement of native biodiversity.

Fig. 4.1. A tree plantation provides many of the same ecosystem services as a natural forest but is of much less value for maintaining biodiversity. Credit: J. Kelly.

Furthermore, not all aspects of nature are benevolent (McCauley 2006). Fires destroy timber resources, grizzly bears kill people, beavers flood roads, and so on. Yet, fires, bears, and beavers are all components of natural ecosystems. Emphasizing only the economically positive aspects of nature leads to a simplification of ecological systems, to the detriment of biodiversity overall. We have plenty of historical examples, such as the cascading ecological effects of past predator control programs (Hebblewhite et al. 2005).

Although the debate over ecosystem services continues, we are not faced with an impasse. It is not an all-or-nothing situation, but a matter of deciding how and where the concept applies. The logical home for ecosystem services in Canada is the agricultural zone and areas of high human population density. It is here that credible markets exist for services related to water quality, flood control, pollination, recreation, and so forth. And it is here, in these highly altered systems, that ecosystem services and biodiversity conservation are most clearly aligned.

For example, providing incentives for reducing agricultural fertilizer and pesticide runoff can improve water quality, benefiting both humans and wild species in the region.

The prospects for applying the ecosystem services approach to Canada's hinterlands are more limited because there are few population centres to serve as markets. For example, the extensive peatlands of the Hudson Bay Lowlands filter vast quantities of water, but this filtering service has little economic value because there are few people to make use of it. The water simply drains untouched into Hudson Bay. There may still be a market for some non-extractive services, such as carbon storage. But, in general, the justification for maintaining the ecological integrity of remote landscapes will continue to be based mainly on intrinsic biodiversity values and not on utility values.

In conclusion, the ecosystem services concept can help advance certain aspects of conservation by drawing attention to the unrecognized economic benefits of nature. But the concept is not a substitute or alternative to biodiversity conservation. These are two parallel processes that overlap to varying degrees depending on the spatial context. There will be many cases where conserving economically important ecosystem services will benefit biodiversity. But overall, maintaining biodiversity will require conservation efforts that exceed what can be justified on purely economic grounds. Thus, conservation will continue to entail hard political choices that pit economic development against the intrinsic and intangible values of biodiversity held by a large proportion of Canadians.

Box 4.2. Conflating Biodiversity Conservation and Ecosystem Services in Ontario

It is not uncommon to see ecosystem services advanced as a rationale for protecting biodiversity. Ontario's biodiversity strategy (OBC 2011) provides an example:

> Conserving Ontario's biodiversity is very important because healthy ecosystems sustain healthy people and a healthy economy. We derive benefits from the ecosystem services provided by biodiversity including food, fibre and medicines, clean air and water and outdoor recreation that nourishes our physical and mental health. Ontario's biodiversity also has inherent value and deserves to be recognized, appreciated and conserved for its own sake. (p. ii)

Though the intrinsic value of biodiversity is mentioned at the end of this statement, the main message is that we are protecting biodiversity for its utilitarian benefits. Highlighting the utilitarian benefits of conservation helps to build support and is politically appealing. However, portraying these benefits as the main objective of biodiversity conservation is misleading and can lead to unintended consequences. The provision of food, fibre, clean water, and so forth demands a different management approach than what is required for maintaining species. Therefore, it is best to advance ecosystem services and the conservation of biodiversity as parallel objectives, rather than presenting biodiversity as the means to an end.

The Role of Science in Conservation Practice

The scientific enterprise involves a combination of observation and structured inference. We may, for example, measure the density of Arctic grayling in selected streams to determine their status and temporal trends. If we also measure ancillary variables, such as water temperature, water quality, and intensity of angling, we may begin to infer causal relationships. In time, such causal relationships can be assembled into a conceptual model that explains how the overall grayling system works. If the model components are sufficiently robust, the model can be used to make quantitative predictions about how the system is likely to respond to specific management actions.

The general process of observation, inference, and model building is not unique to science. Each of us builds our own mental model of how the world works based on our life experience and what we learn from others. For example, anglers use their personal models to determine the best spots to fish. This approach also underlies Indigenous traditional ecological knowledge.

What differentiates and ultimately defines science is the methodology used to guide the process. Established techniques exist for designing studies, making observations, and analyzing data. These methods optimize the utility and reliability of observations and allow uncertainty to be quantified (Mills and Clark 2001). Furthermore, conclusions are treated as hypotheses, not facts, and are subjected to peer challenge and continued testing. Scientific information is by no means infallible, but science is the best approach available for separating true relationships from chance associations and developing a robust understanding of how the natural world works.

As discussed in Chapter 1, science supports conservation by informing the decision-making processes that determine conservation actions. The specific contributions that science makes vary according to the stage of the process (Mills and Clark 2001; Barbour et al. 2008; Arlettaz et al. 2010):

- **Setting objectives.** Objectives define what we would like to achieve in a policy or planning setting. They express what is important to us, implying an origin in societal values rather than science. The role of science is to identify threats to our values and to quantify the level of risk, thereby motivating action. Thus, science plays an important role in agenda setting.
- **Identifying options.** Most conservation problems are a consequence of human activity, in one form or another. Therefore, conservation usually entails searching for alternative, more benign, approaches for interacting with nature. Science contributes to this process by elucidating the causes of ecological problems and identifying potential remedies.
- **Making a decision**. The last step in the decision-making process is formulating a management response, which entails assessing the available management options and selecting the best. In conservation applications, this is typically a political process because it involves judgments about societal risk tolerance and trade-offs among values. Science supports the process by predicting outcomes under the different management options and by quantifying uncertainties.
- **Monitoring and learning**. Scientifically rigorous monitoring and research can reduce management uncertainties and provide the information needed to evaluate the effectiveness of past decisions. This changes

decision making from a linear process to a cyclical process of continuous improvement. Monitoring also serves to identify new and evolving problems.

In subsequent chapters, we will examine how these various aspects of science are applied in practice. Real-world applications are messy and often fall short of the ideal. Therefore, we will pay close attention to the differences between theory and practice as we proceed.

Policy-Relevant Research

By the early 2000s, it was increasingly apparent that a disconnect existed between the academic study of conservation and the "on-the-ground" practice of conservation. The scientific literature was biased to conservation theory and not well suited to real-world applications (Knight et al. 2008). Commentators referred to this as an "**implementation crisis**" (Biggs et al. 2011).

The disconnect between research and practice reflected a reluctance by academic scientists to stray very far from their professional comfort zone. In the words of Knight et al. (2006 p. 410), "Few academic conservation planners regularly climb down from their ivory towers to get their shoes muddy in the messy political trenches, where conservation actually takes place." In part, this was a consequence of the reward structures within academic institutions, which were geared toward publication in high-profile journals rather than the support of "hands-on" conservation (Hallett et al. 2017). Scientific journals, for their part, placed a priority on novelty and were little concerned with the particulars of specific management applications. Finally, the reality was that problems that were interesting were not always important, and problems that were important were not always interesting (Cook et al. 2013). Moreover, many researchers had a personal aversion to the policy arena, which they perceived as a foreign and unfriendly landscape.

In recent years, an effort has been made to build support for **policy-relevant research**, also referred to as **translational ecology**, and to develop principles and guidelines for its implementation (Reed et al. 2014; Enquist et al. 2017; Hallett et al. 2017). Policy-relevant research is science with a mission. Success is not measured in terms of where a paper is published or how many citations it receives but in its utility in supporting practical conservation decision making and action. One might argue that this is what conservation biology has always been about. But the implementation crisis suggests that, in the past, this has been more of an aspiration than a reality.

The foundation of policy-relevant research is direct communication with other participants in the decision-making process (Biggs et al. 2011; Reed et al. 2014). This communication needs to be "a two-way process based on effective relationships rather than on simply telling" (Forbes 2011, p. 221). Applied researchers also should understand how policy and planning decisions are made.

A solid grasp of the social dimensions of the issue at hand is another prerequisite (Game et al. 2015). This requires engagement at the local level. What are the main concerns? What perspectives do key stakeholders hold? What are the major points of conflict and barriers to implementation? Who will implement the decisions, and what constraints do they operate under?

Policy-relevant research also requires careful attention to study design. Studies should be situated within a broader decision-making framework and orientated toward implementation (Knight et al. 2008). The key attributes of policy-relevant research are salience, credibility, and legitimacy (Cook et al. 2013). **Salience** assures that the research is relevant and timely and that the format, timing, and resolution are appropriate. **Credibility** assures that a study is perceived as authoritative, believable, and trusted because of a transparent and robust scientific process. **Legitimacy** assures that the research process takes account of the values, concerns, and interests of all relevant actors, as well as practical constraints on decision making such as economic cost and existing policy.

Another important aspect of study design is determining how far to extend the research into the domain of social decision making. In the past, the tendency has been to focus on the ecological aspects of a problem and then pass the baton. Reserve design is a prominent example. Countless reserve designs have been generated that represent optimal reserve configurations from a purely biocentric perspective. These are dutifully passed on to government decision makers who often dismiss the recommendations on the grounds of impracticability (Knight et al. 2008).

Better conservation outcomes can be achieved if socio-economic trade-offs are incorporated into the reserve design process (see Chapter 8). By doing so, reserve designs can be identified that achieve stated conservation objectives while minimizing conflict with other values. Such designs are more likely to be implemented than designs that focus only on the biotic dimension. Doing this effectively requires collaboration between researchers, land managers, and stakeholders. The same principles apply to other forms of conservation research.

Policy-relevant research is most challenging when existing policy is itself part of the problem (Karr 2006). In such cases, research into policy alternatives may be of greater benefit to biodiversity than supporting the implementation of an existing policy. However, research of this nature may be disregarded by decision makers, at least in the short term, because of a perceived lack of alignment and legitimacy. Thus, the dilemma facing researchers is deciding which course of action—supporting an existing policy or challenging it—will be of greatest benefit to conservation. This is part of the agenda-setting role of science.

Because policy-relevant research has an applied focus, it is possible, and indeed imperative, to learn from experience. Ehrenfeld (2000) states:

> We must give up the self-serving belief that an increase in our scientific knowledge by itself will always move us toward effective conservation. To help identify conservation strategies that work, conservation biology must close critical feedback loops by emulating medicine and regularly monitoring the effectiveness of its research and recommendations. (p. 105)

The final component of policy-relevant research is knowledge transfer (Reed et al. 2014). Simply publishing findings in research journals is not sufficient. Few managers have time to read and synthesize the relevant primary literature (Pullin et al. 2004). Also, it cannot be assumed that the facts speak for themselves. Instead, the scientists who conduct the research are in the best position to interpret the findings and recommend how they apply to specific policy or management decisions (Noss 2007).

Knowledge transfer to decision makers is achieved by channelling research findings into review articles and summaries and into decision support systems (Dicks et al. 2014). It is also achieved through direct collaboration in decision-making processes (Enquist et al. 2017). Researchers should make an effort to frame the information in a way that is understandable and actionable (Weber and Word 2001; Forbes 2011). Aspects of the research that impinge directly on management decisions and implementation issues should be emphasized. Finally, outreach efforts should target not only government decision makers, but other actors in the policy process, including ENGOs, industry, and other stakeholders (Lach et al. 2003). In Chapter 10, we will discuss how this can be accomplished within a structured decision-making framework.

In summary, policy-relevant research provides a vital bridge between basic biological science, conservation theory, and conservation as it is practiced in the field. Policy-relevant research does not replace basic science and theory; it is an extension of them. Different skill sets are involved, so a degree of specialization is to be expected. Moreover, individual researchers may emphasize different roles over the course of their careers, as their knowledge and interests evolve.

Challenges

Making science relevant to decision makers is not the only challenge in its application to conservation. Capacity constraints are a major limitation, both in terms of funding and the availability of researchers with relevant expertise. Another problem is that ecological research is inherently time-consuming, which means that knowledge gaps related to pressing management issues cannot be addressed quickly (Mills and Clark 2001). In practice, decision making must often proceed with incomplete knowledge. The value of research tends to be realized over the longer term, in successive iterations of the policy and planning cycle. Finally, because ecological systems are highly complex, research results are usually accompanied by caveats and contingencies. This can frustrate decision makers seeking straightforward answers to their problems.

Also, though it may seem obvious that incorporating scientific information produces better decisions than simply muddling through, it is not always welcome. Science may be rebuffed when it draws unwelcome attention to policy failures or is seen as a challenge to the status quo (Wilson 1998). As we saw in Chapter 3, political leaders and government bureaucrats often find change threatening and may resist it. As a result, scientific information concerning ecological threats is sometimes willfully disregarded and management solutions ignored, to the frustration of conservation practitioners.

In the worst case, governments may actively discourage the creation and dissemination of scientific information or attempt to intimidate or discredit individuals and organizations engaged in generating it (Carroll et al. 2017). In Canada, we saw this happen with the Harper government's "War on Science," and it was also a key feature of the Trump administration in the US (Turner 2013).

We are now also witnessing the rise of an anti-science movement among factions of the public, with widespread skepticism of climate change and vaccination as prominent examples. The causes of this movement are manifold and complex; however, one of the main factors is increased polarization of society, abetted by media fragmentation (Carmichael et al. 2017). What you believe increasingly depends on which "tribe" you belong to (Hayhoe and Schwartz 2017). The replacement of reasoned debate with arguments over "alternative facts" is extremely retrogressive and bodes poorly for conservation and policy development in general.

Citizen Science

Field research by academic and government scientists has been the source of most of the quantitative data used to support conservation. However, this is not the only approach available. Over the past few years there has been an explosion in citizen science, largely thanks to new smartphone apps linked to online databases (McKinley et al. 2017). Through the efforts of volunteer naturalists, millions of species observations are being added to online databases each year.

Citizen science is not new—observing and documenting animals and plants is something that naturalists have always done. What's new is the enormous increase in participants and the rapid, widespread sharing of information made possible by the Internet. The quality of observations has also increased, through the inclusion of photographs, sound recordings, automatic time stamps, and GPS locations.

The Contribution of Citizen Science

Citizen science is the practice of engaging the public in scientific projects that produce reliable quantitative information usable by scientists, decision-makers, and the public (McKinley et al. 2017). In the field of conservation, most citizen science projects focus on broad-scale species monitoring. These projects provide information on species abundance, species distribution, migratory patterns, and the timing of natural processes such as flowering. Some projects focus on monitoring aquatic and terrestrial habitat quality. In addition, researchers and resource managers sometimes recruit volunteers to help collect data for specific research studies.

The core strength of citizen science is the large number of observers it engages. This complements the main weakness of conventional science, which is limited capacity. Scientists are very good at collecting high-quality data, but there just aren't enough of them. Placing literally millions of additional observers in the field makes a tremendous difference in what can be achieved. Through citizen science, we can monitor across vast spatial scales and over extended time frames. We can also track uncommon species and species that are generally overlooked through conventional monitoring programs.

Structured Projects

Citizen science projects come in two main forms: structured and unstructured (Callaghan et al. 2018). Structured projects are established for a defined purpose. Typically, the organization or individual that initiates the project also leads the analysis of the data and the application of the findings. Structured projects are characterized by well-defined protocols that observers must follow. These protocols define what is to be studied as well as standardized methods for making observations.

The Breeding Bird Survey provides a good example of a structure project. Each spring, experienced volunteers count birds along a series of fixed survey routes using a standardized observation protocol (Fig. 4.2). Observers stop every 800 m along their assigned route and count all birds they see or hear over three minutes. The count begins 30 minutes before sunrise on a suitable day between May 28 and July 7.

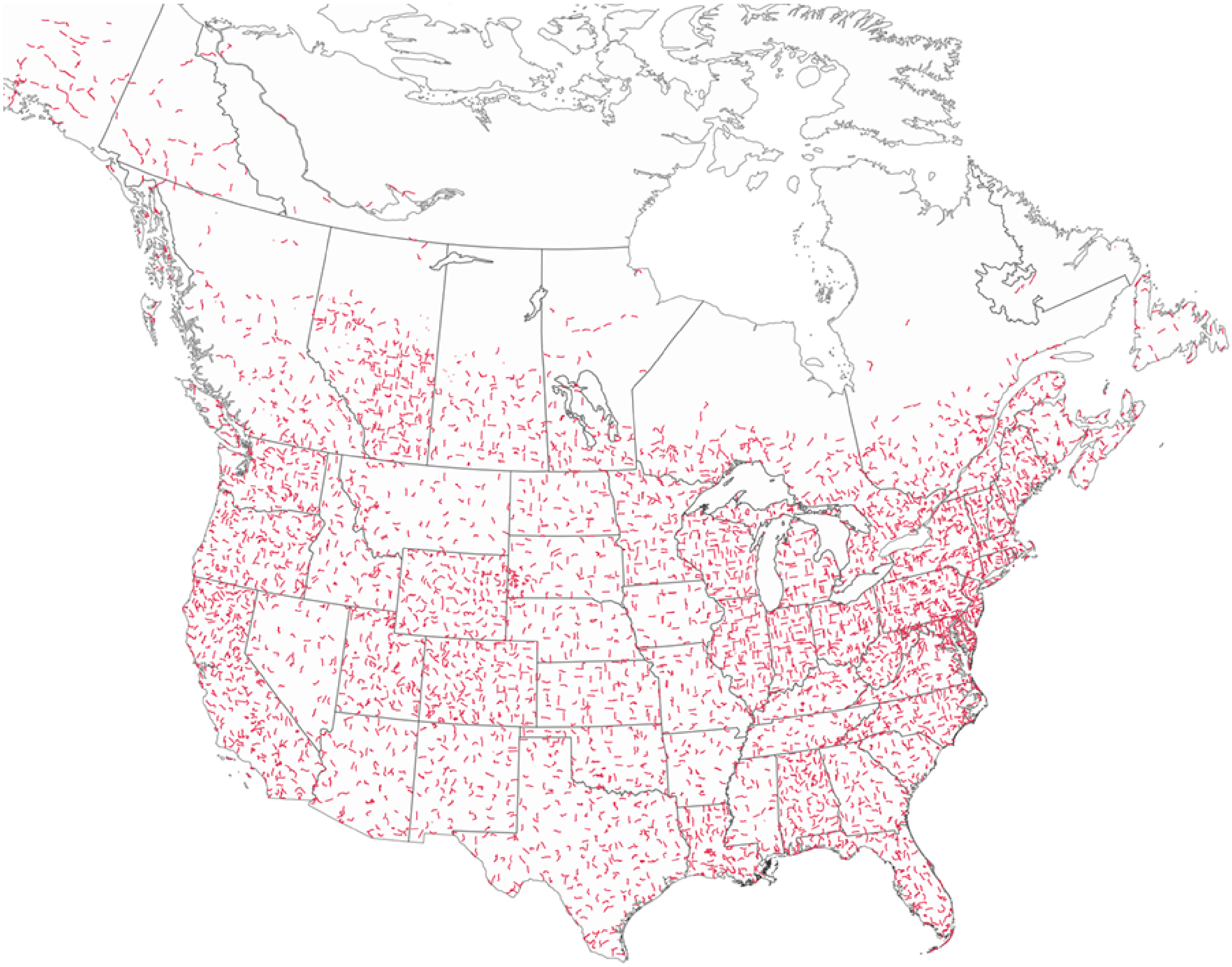


Fig. 4.2. There are approximately 3,700 Breeding Bird Survey routes in Canada and the US. Each roughly 40-km long route is made up of 50 stops where experienced volunteers identify and count all birds seen or heard in a 3-minute period.

The Canadian Wildlife Service oversees the program in Canada and analyzes the data and generates regular reports on the state of birds in Canada. Data generated by the Breeding Bird Survey have contributed to over 500 peer-reviewed papers and the data are used in many policy decisions, including species listings. No better dataset exists for tracking wide-scale changes in bird abundance over time.

Unstructured Projects

Unstructured projects are more open-ended. Generally, the intent is to build a database of observations without explicitly defining what the data will be used for. Projects may impose some requirements on observers, such as requiring a photograph or limiting which species are to be included. But participants are otherwise free to make observations whenever and wherever they please.

Unstructured projects can make an important contribution to conservation science despite the opportunistic nature of the observations. In the case of iNaturalist and eBird, the two largest projects, size is a key factor—these projects are global in scope and have *millions* of participants. Never in the history of science has so much information been collected about so many species. The large number of observations permits meaningful insights to be made, despite biases and variability among observers.

iNaturalist

iNaturalist is an online platform and associated smartphone app used to record citizen science observations (Fig. 4.3). The observations are unstructured: participants can submit sightings of any species they wish at any time of the year. Most observations are made using the smartphone app and consist of a photograph or sound recording of an individual species together with a time stamp and GPS location. Participants are prompted for a preliminary species identification, which is later verified by the iNaturalist community of naturalists. After the species identification has been verified, the observation is labelled as "research grade" and considered suitable for statistical analysis.

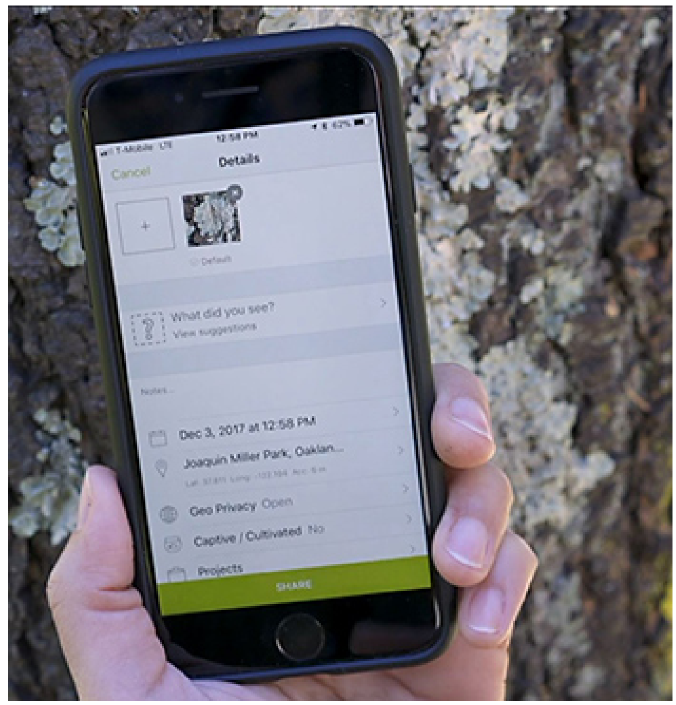


Fig. 4.3. Smartphone apps make it easy to engage in citizen science, which has led to a tremendous growth in participation in recent years.

As of 2023, the iNaturalist database contains over 130 million observations of 420,700 species. Within Canada, there have been 9.8 million observations of 34,919 species. The database is available to everyone at no charge and can be filtered by region, species, and date.

The strength of iNaturalist is that it generates an enormous number of observations over a wide spatial and temporal extent. As such, it excels at describing species distributions, especially of species that are overlooked or difficult to monitor using conventional methods (Fig. 4.4). It is less suited to estimating species abundance because the sampling effort is unknown and because observers do not provide a complete inventory of species at a given location. That said, statistical methods are being developed to maximize the value of such "presence only" datasets (e.g., Fithian et al. 2014). Users of the data also have to account for several forms of observation bias, such as proximity to population centres and the ease of photographing a given species (Callaghan et al. 2021, Feldman et al. 2021).

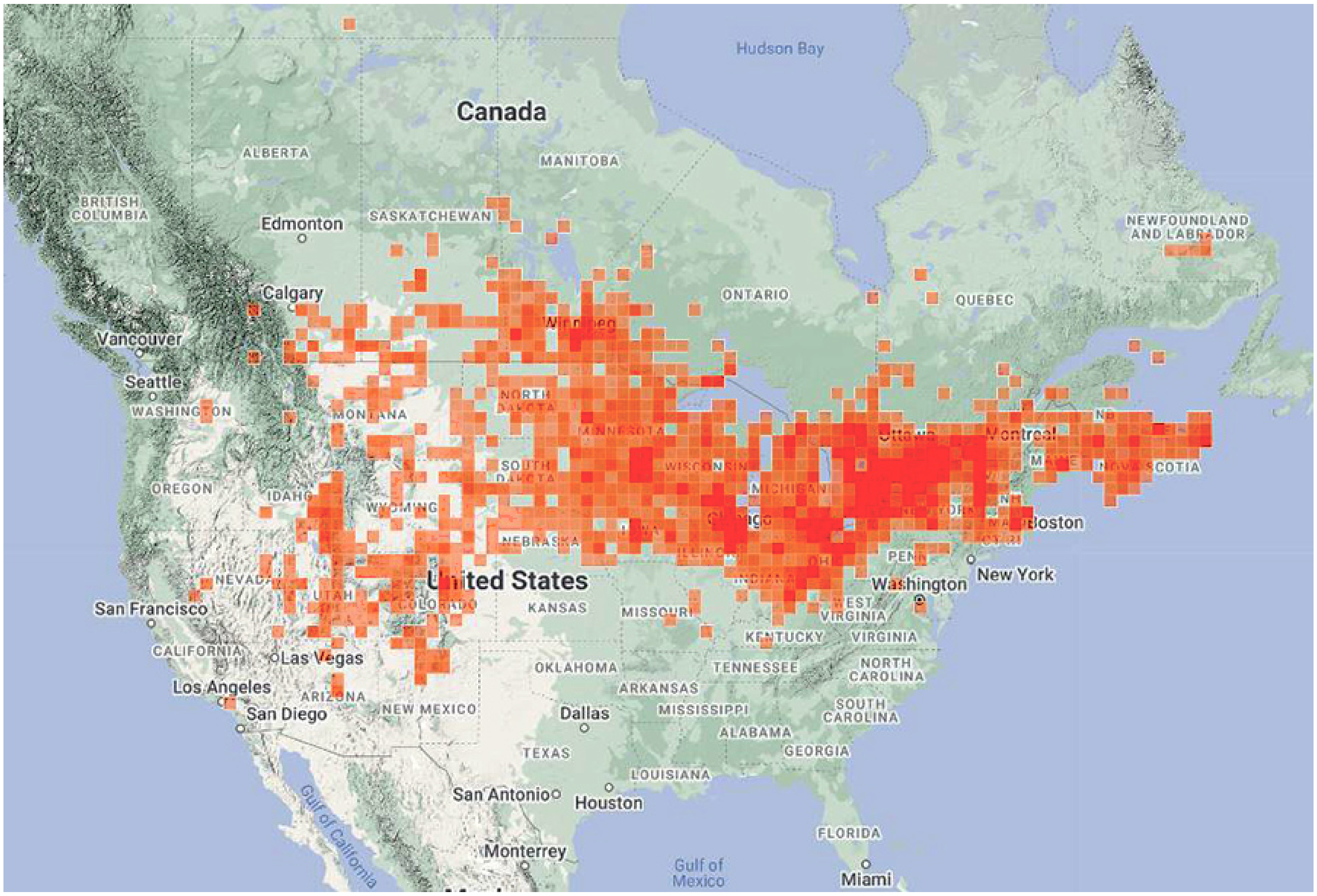


Fig. 4.4. The distribution of leopard frogs based on iNaturalist observations. Darker shades of red indicate a higher number of observations. Species distribution maps like this can easily be downloaded from the iNaturalist website.

eBird

Like iNaturalist, eBird is an online platform with an associated smartphone app. Since its release in 2002, it has quickly become a favoured resource for birders around the globe. As of 2023, 820,000 birders have contributed more than *1.3 billion* observations to the database (eBird 2023).

eBird uses a checklist approach, carried over from the pre-digital era. The intent is to list every bird seen or heard on an outing, creating a complete census of a site. The eBird app facilitates the recordkeeping process while also tracking the duration of the outing (a measure of sampling effort) and the GPS location. The app also flags unexpected sightings to minimize reporting errors.

Because eBird checklists are contributed throughout the year, and at the global scale, they provide unparalleled insight into bird distributions and migratory patterns (Fig. 4.5). Moreover, the checklist approach provides a measure of both species presence and absence at each survey location, as well as a measure of observer effort, permitting the estimation of relative abundance and trends over time (Fink et al. 2020). Like iNaturalist, observation biases exist and must be taken into account (Feldman et al. 2021).

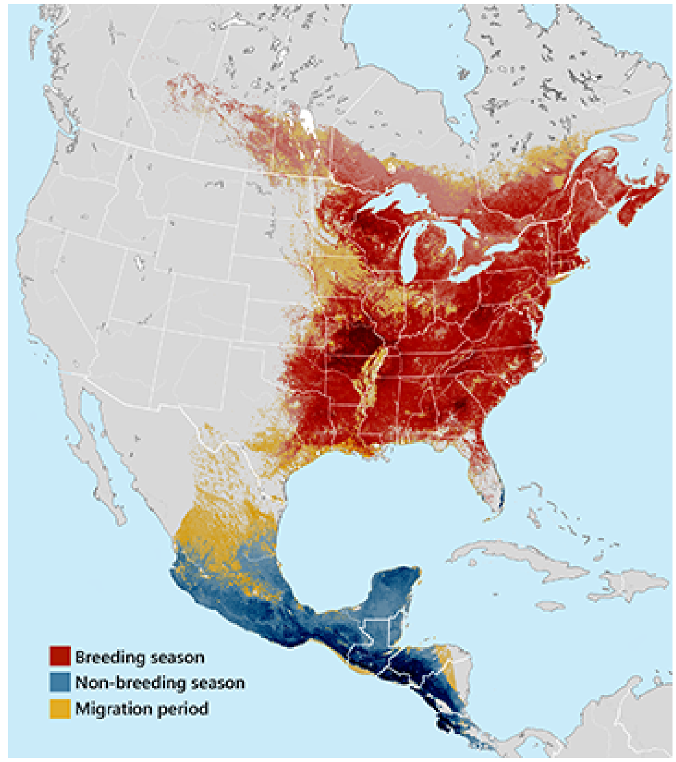


Fig. 4.5. eBird seasonal distribution map for the ruby-throated hummingbird. Darker shades indicate higher abundance.

Applying Citizen Science to Conservation

With the rising profile of citizen science in recent years, the number of projects has greatly increased. The value of these projects for supporting conservation is variable and can be assessed on six main criteria (AEP 2020):

- **Accessibility**. Projects should facilitate the broad, public sharing of information (at no charge). Projects that maintain private databases are less desirable.
- **Data products**. Projects with the capacity and technical expertise to analyze the incoming observations and provide data products such as distribution maps and summary reports are preferable to projects that just store the data.
- **Spatial and temporal scope**. Species do not respect administrative boundaries. Therefore, all else being equal, a national or global project is preferable to one that is local in scope. Similarly, projects that collect observations all year long are preferred over projects that provide only a snapshot at a particular time of year.
- **Data quality**. Projects that have standardized protocols for making observations generate higher-quality data than projects that do not. So do projects that check the validity of species identifications submitted by observers. High-quality data facilitates the ability of researchers to draw meaningful insights from the observations.
- **Application**. Projects that have the attention of researchers and resource managers are preferred over those that do not. The more direct the connection between observations and their management application, the better.
- **Data security**. It is an unfortunate reality that citizen science initiatives sometimes shut down after a period of time, typically because of a lack of funding, a change in an organization's priorities, or the loss of key people. Some projects have also lost data (mainly in the era before cloud-based storage). Therefore, consideration should be given to the long-term viability of the project and its ability to store data reliably.

Despite the inherent benefits of large projects, the value of small, local projects should not be discounted. A small project may have very high conservation value if it is focused on gathering data for a specific local application. In other words, purpose and application are overriding factors.

To be clear, citizen science is not a competitor to conventional research. Well-designed research projects will always be the best approach for generating high-quality information with high efficiency. But there is a limited supply of researchers and project funding, which severely constrains what can be accomplished through conventional field studies. Consequently, conservation researchers and practitioners are increasingly turning to citizen science to augment (not replace) the information available from conventional studies (Adde et al. 2021). For example, in 2022 alone, 1,775 academics downloaded the eBird dataset for analysis and published 160 peer-reviewed papers using this information (eBird 2023). Citizen science is also increasingly used by managers to inform species management decisions (McKinley et al. 2017, Ruiz-Gutierrez et al. 2021).

The Role of Conservation Practitioners

We will now shift our focus to the role of conservation practitioners, loosely defined as individuals with some form of conservation expertise working on applied conservation issues. This is a diverse group that operates at the interface between science and policy. Some conduct applied research, some engage in policy development and planning, and some oversee the implementation of conservation measures. Many serve in more than one capacity. Notably, none of the three modes of activity are exclusive to conservation practitioners (Fig. 4.6). What differentiates this group is their primary orientation toward achieving conservation outcomes.

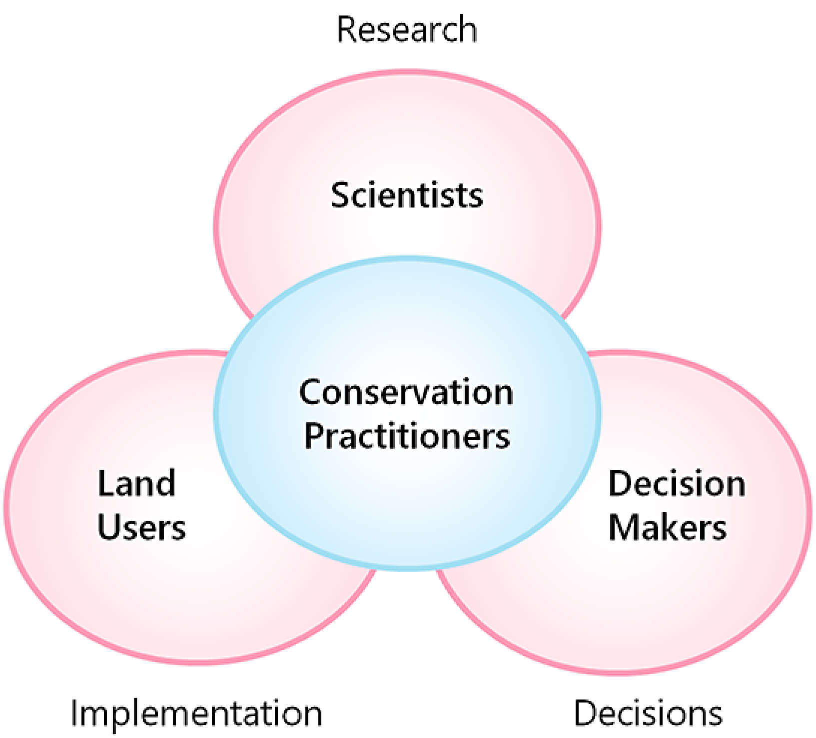


Fig. 4.6. Conservation practitioners have three main roles: research, decision making, and the implementation of conservation measures. What differentiates conservation practitioners from others working in similar capacities is their primary orientation towards achieving conservation outcomes.

As researchers, conservation practitioners are distinguished from other scientists by their focus on applied questions related to the implementation of conservation in specific settings. Thus, not all biologists consider themselves conservation practitioners, even though basic science does support conservation indirectly. In practice, the line between basic and applied research is blurred, and it is best to think of these terms as the poles of a spectrum. Furthermore, many scientists conduct both basic and applied research.

In terms of policy and planning, conservation practitioners play a critical role in synthesizing scientific knowledge about conservation and bringing it to bear in decision-making processes. Research findings locked within the pages of a journal are of no practical value until they are mobilized. How this is done depends on the type of decision being made.

At the base of the decision hierarchy are operational decisions related to the implementation of higher-level plans. For example, once a decision has been made to install a wildlife road crossing, operational decisions are required to determine the optimal location and size. These sorts of decisions are mainly technical, requiring the direct application of scientific expertise. Conservation practitioners typically have primary responsibility for decisions at this level.

Moving up the decision hierarchy, the social aspects of conservation decisions begin to predominate. This is because, as the scope of a decision broadens, more stakeholders are affected and trade-offs with competing values require greater consideration. In such cases, conservation practitioners are more likely to serve as scientific advisors than decision makers. At the highest levels, decisions are made by elected officials, even if the focus is on conservation. For example, while conservation practitioners informed the development of Canada's *Species at Risk Act*, the core decisions were made by the federal cabinet.

As for implementing conservation measures, this mostly entails managing the activities of land users (both resource companies and individuals) through regulations, guidelines, and incentives. Here, conservation practitioners typically serve in an oversight role, advancing conservation through education, outreach, and the enforcement of regulations. Some conservation initiatives also feature a "hands-on" component that requires the direct involvement of conservation practitioners. Examples include habitat restoration initiatives, species reintroduction projects, and habitat management programs.

We will examine these roles in a variety of applied contexts in subsequent chapters. In Chapter 12 we will discuss skills training and the determinants of success.

Institutional Connections

Within governments, conservation practitioners are typically employed in biologist and resource management positions. Many work in fish and wildlife departments, where their focus is on species at risk and game species. Others work in forestry departments and parks departments, where they engage in conservation at both the species and ecosystem level. The range of activities of government-based practitioners is broad and includes policy development, planning, implementation, and sometimes research. These practitioners may serve either as technical advisors or decision makers, depending on the application.

As would be expected, conservation practitioners within the academic community focus mainly on research and teaching. Research programs are often structured around questions that arise out of policy and planning processes. Governments and industry, for their part, may provide funding, and over time, mutually beneficial relationships sometimes develop. Some scientists also engage in policy and planning more directly, by giving presentations and media interviews about conservation issues, providing expert advice to policymakers, and serving on planning bodies.

ENGOs are another common home for conservation practitioners. Again, the range of activities they engage in is broad. In some organizations, such as Ducks Unlimited and the land trusts, the emphasis is on directly implementing conservation measures. Other groups engage mainly in conservation planning and policy through outreach, advocacy, and participation in planning initiatives. Internally, conservation practitioners direct their group's conservation programs and provide the scientific foundation for position statements and policy recommendations.

Some groups, such as the Wildlife Conservation Society and Bird Studies Canada, emphasize applied research and publish studies in peer-reviewed journals.

Within industry, conservation practitioners are mostly employed by forestry companies as biologists and foresters. These individuals have lead responsibility for achieving the ecological objectives that forestry companies have committed to under sustainable forest management (both at the species level and ecosystem level). Their responsibilities and ability to influence harvest practices vary considerably from company to company. In some cases, funding for collaborative research is available and there is high-level support for trying new approaches. Other companies prefer to maintain the status quo.

Other industrial sectors, particularly oil and gas and mining, tend to obtain the conservation expertise they need by hiring ecological consultants on an ad hoc basis. Governments also frequently use ecological consultants, particularly now that progressive downsizing has reduced internal capacity. These ecological consultants form another pool of conservation practitioners. Much of the work these consultants do revolves around environmental assessments of large industrial projects and the associated regulatory filings, mitigation plans, and reclamation plans. Consultants with expertise and interest in the broader application of conservation science also exist, and projects and planning initiatives that require their services arise on occasion.

The Controversy over Advocacy

Advocacy is the act of publicly supporting or arguing in favour of a cause, an idea, or a policy. This may seem to have little to do with science, which is concerned with the objective study of the natural world. But in the application of science to policy problems, the line between providing scientific advice and recommending a particular course of action may be difficult to discern (Horton et al. 2016). Whether or not this is a problem has been the subject of a long-standing debate in the conservation literature (Mills 2000; Lackey 2007; Noss 2007; Nelson and Vucetich 2009). The particulars of this debate provide additional insight into the role of science in conservation.

Opponents of advocacy argue that it threatens effective decision making (Lackey 2007). Technical experts command a privileged position because decision makers look to them for reliable advice about the nature of policy issues and what to do about them (Meyer et al. 2010). The danger of advocacy is that it threatens this relationship (Lackey 2007). If experts are perceived to be advocating for a particular outcome, decision makers may question whether the advice they are receiving is truly objective and reliable. Experts may then find their advice lumped in with the value-laden views of stakeholders, to the detriment of the entire process.

Objectivity is also needed to resist the politicization of science by stakeholders who often use science-based arguments to bolster their position in public debates. Stakeholders may even seek to fund research designed to support their specific views (Jacques et al. 2008). This can tarnish the reputation of science and reduce its credibility if it leads to selective reporting of findings and other forms of distortion.

Given these concerns, the opponents of advocacy suggest that technical experts have an obligation to remain objective, neutral conveyors of scientific information (Lackey 2007). Experts should “describe empirically the way things are, not the way we think they ought to be” (Barbour et al. 2008, p. 564). Lackey (2007) provides this advice:

> Be clear, be candid, be brutally frank, but be policy neutral when providing science to the public, policymakers, and others. … Often I hear or read in scientific discourse words such as degradation, improvement, good, and poor. Such value-laden words should not be used to convey scientific information because they imply a preferred ecological state, a desired condition, a benchmark, or a preferred class of policy options. Doing so is not science, it is policy advocacy. Subtle, perhaps unintentional, but it is still policy advocacy. … Why use them unless you are conveying the impression that one particular condition is preferred policy wise? A forest that has been clearcut is degraded habitat from the perspective of spotted owls and red tree voles, but it is improved habitat from the perspective of other species such as white-crowned sparrows and black-tailed deer. The science is exactly the same, only the policy context differs. The appropriate science words are, for example, change, increase, or decrease. (p. 14)

The proponents of advocacy assert that Lackey's perspective on the role of science is fundamentally at odds with the tenets of conservation biology and applied science in general (Noss 2007; Horton et al. 2016). There is more to science than the description of natural processes; it is also a tool for achieving specific goals. Indeed, a considerable proportion of all scientific inquiry is goal oriented. For example, if we conduct a study to determine the most effective method of preventing heart disease, we are conducting goal-directed research. The results will naturally be expressed in the context of that goal: less heart disease is labelled as a positive outcome, not simply a change. It is no different for research intended to support the conservation of biodiversity.

The credibility of applied research is maintained by stating the research goals upfront and by rigorous application of the scientific method, buttressed by institutional standards and oversight (Ehrlich 2000). Concerns about objectivity apply to how studies are conducted and reported, not to the purpose of the research. Sometimes, maintaining objectivity means resisting pressure from employers or funders. More generally, it means guarding against personal biases and preconceived ideas. In the words of the physicist Richard Feynman, "The first principle [of science] is that you must not fool yourself, and you are the easiest person to fool" (Feynman 1985, p. 313). This requires a willingness to question your own assumptions, as well as those of your field of study, and to change your opinion when compelling evidence suggests you should.

Credibility also demands honesty and openness (Meyer et al. 2010; Horton et al. 2016). Rather than hiding biases, or pretending they do not exist, they should be openly stated. In addition, researchers should speak to what they know and acknowledge when they are moving beyond their area of expertise. This includes clearly distinguishing between data, inference, and informed speculation, which are usually perceived as one and the same by external audiences (Brussard and Tull 2007). There should also be an openness about uncertainties and persistence in telling policymakers what they can reasonably expect from science, and what they cannot.

In the final analysis, both the critics and proponents of advocacy present valid arguments. It is quite possible to conduct applied conservation research that is accepted as reliable scientific information by decision makers. However, overt advocacy, particularly the promotion of specific management choices, will quickly change how the information is perceived. In practice, practitioners must make a choice. They can serve as technical experts or they can serve as stakeholders promoting conservation as a value position. But they rarely can do both at once.

Conservation practitioners also need to consider the mandate and expectations of the organizations they work for. In many organizations, particularly within government, conservation practitioners are expected to adhere to their assigned roles (Steel et al. 2004). Strong advocacy efforts by conservation practitioners may be viewed by their superiors as a challenge to their authority (Lach et al. 2003). This is especially true when statements are made publicly. Conservation advocacy may also conflict with other government mandates, creating internal discord.

The final aspect of advocacy that bears mention is advocacy on behalf of science itself. As noted earlier, science is coming under assault, to the detriment of reasoned debate and good decision making. Conservation practitioners should join with others in the scientific community to help the public understand the importance of science and evidence-based decision making (Carroll et al. 2017). For example, in April 2017, more than one million science supporters in 600 cities across the world marched in the streets to raise the profile of science and champion its importance to society (Fig. 4.7). Such efforts need to be followed up with individual everyday efforts to encourage support for science and effective decision making.

Fig. 4.7. The March for Science took place on April 22, 2017, with over one million scientists and supporters participating in 600 cities around the world. The march was organized to draw attention to the importance of evidence-based decision making and the need for governments to support research. Credit: Becker.

CHAPTER V

THREATS TO BIODIVERSITY

Threats to Biodiversity

Patterns of Decline

In this chapter, we will examine the major threats to biodiversity, setting the stage for our discussion of specific conservation actions in the chapters that follow. We will begin by examining current patterns of decline and then turn to the causes, which vary by region. Climate change, which is expected to profoundly alter species and ecosystem distributions in coming decades, will be discussed separately in Chapter 9.

In conservation, a threat is a process that has the potential to harm biodiversity—which is understood to mean a change from its natural state. For now, we will use the preindustrial landscape as our reference point, allowing us to identify threats and the level of risk they pose. In Chapters 6 and 7 we will discuss the meaning of the natural state in greater detail, in the context of setting conservation objectives. In Chapter 9, we will consider the meaning of the natural state under a changing climate.

Globally, species have undergone massive declines over the past 100 years. It is not only endangered species that are of concern—these are just the tip of the iceberg. A much wider group of species have undergone range contractions and experienced declines in abundance, including many whose status is listed as secure. Vertebrate species have been most studied, and of these, 32% have experienced significant range contractions (Ceballos et al. 2017). Conservation scientists have begun to refer to the current episode of biological loss as earth's sixth mass extinction event (McCallum 2015).

The most comprehensive national-scale assessment of species in Canada is compiled by the Canadian Endangered Species Conservation Council. This multi-governmental body provides an updated report on the general status of Canadian wild species every five years (CESCC 2022). General status assessments integrate the best available information on population trends, distribution, and threats to create a snapshot of each species' status in Canada. Coverage is greatest for vertebrates and vascular plants, which have almost all been assessed. Excluding extirpated and alien (non-native) species, 78% of vertebrates and 74% of vascular plants are currently considered secure (Table 5.1). To be clear, secure means a species has a low risk of extirpation, not that its abundance and range are within natural bounds.

Table 5.1. General status of wild species in Canada in 2020.[1]

Group	Secure	Vulnerable	Imperiled	Critical	Extirpated	Alien	Unranked[2]
Fish	585	51	28	11	4	16	700
Amphibians	28	10	5	3	1	0	0
Reptiles	11	16	9	4	4	2	3
Birds	339	52	22	23	5	11	244
Mammals	133	33	11	11	2	11	22
Plants[3]	2,841	418	293	254	51	1,372	95

[1]Source: CESCC 2022.
[2]Species that occur as infrequent migrants to Canada or for which information is lacking.
[3]Vascular plants, including flowering plants, cone-bearing trees, ferns, and horsetails.

Other taxonomic groups exhibit broadly similar patterns, but assessments are patchy. Lichens, mosses, and insects have received the most attention; however, coverage is still low. Most of the remaining taxonomic groups have received minimal attention. The general status report also lacks information on subspecies status for all groups.

The general assessment report also provides insight into species richness patterns across Canada (CESCC 2022). Species richness is highest in the south, especially southern Ontario, and declines as one moves northward. Richness patterns are also closely tied to landscape diversity. Regions with a high diversity of landforms and climates—particularly BC—have higher levels of biological diversity.

General status assessments only began in 2000, so we must look to other monitoring programs for information on longer-term biodiversity trends. Only a few such programs are available at the national scale. The most extensive is the Breeding Bird Survey, which is a volunteer-based initiative that has been providing national-scale data on the status of birds since the 1970s. Birds are easier to study at broad scales than most other groups, and 76% of species are sufficiently monitored to determine long-term trends (NABCI 2019). A limitation is that coverage for nocturnal birds, wetland specialists, and secretive and rare birds is generally poor. Geographic coverage has improved over time but remains incomplete in northern areas due to limited road access and a paucity of observers. Finally, relating population trends to specific threats is complicated by the migratory behaviour of most birds.

From Breeding Bird Survey data, we can see that population trends have varied widely among bird groups (Fig. 5.1). Aerial insectivores have fared the worst, declining by 59% on average (with many individual species showing even greater declines). Grassland birds have experienced a similar rate of decline. In contrast, waterfowl, raptors, and colonial seabirds have been increasing. Birds of the boreal forest have been stable overall, though some individual species, like the Canada warbler, have undergone major declines.

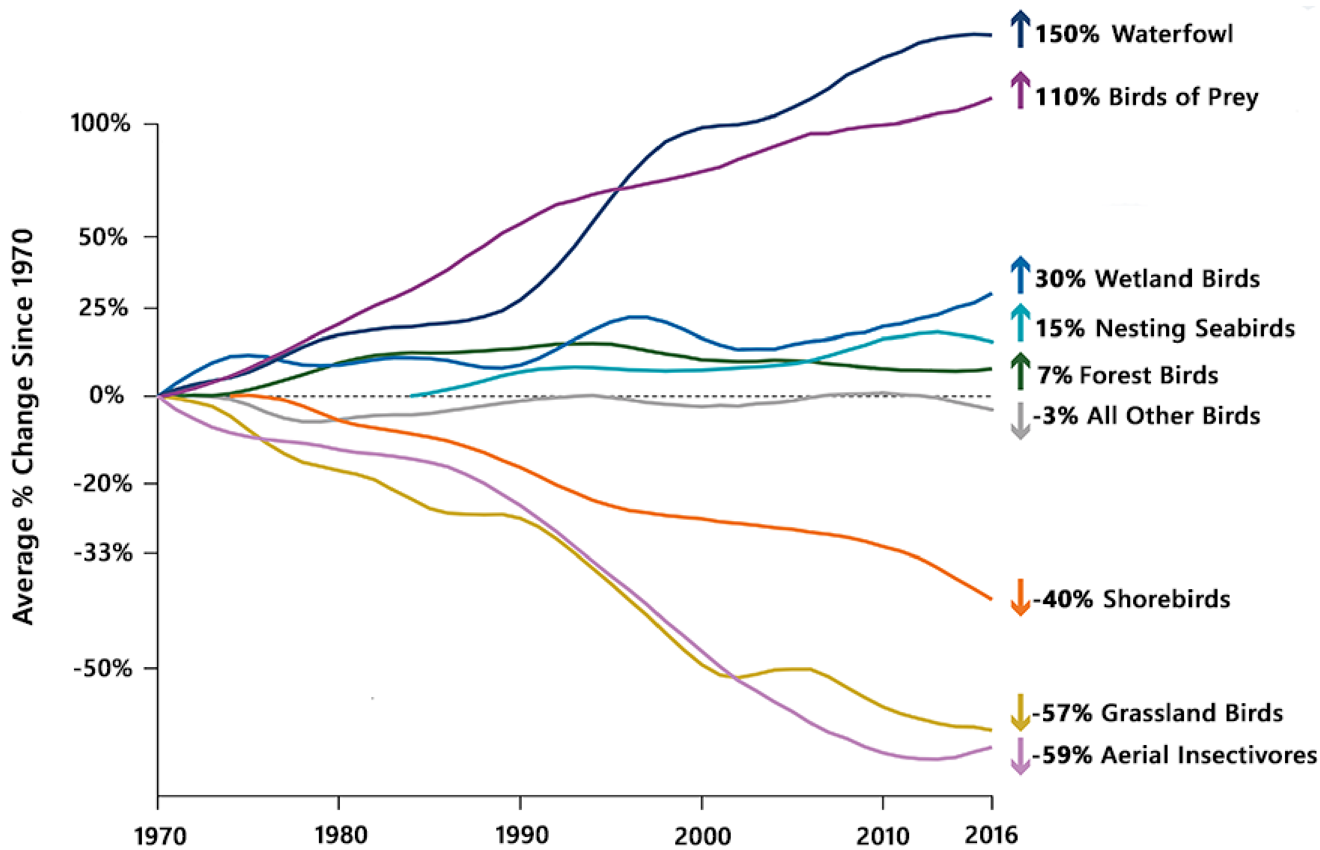


Fig. 5.1. The average change in population size since 1970, by bird group (NABCI 2019.)

In the limited number of other taxa for which trend data are available, the regional patterns are generally consistent with what has been observed for birds (CESCC 2022). Variability among species appears to be a common feature in many taxa, with some species increasing, some decreasing, and others remaining relatively stable over time. Put another way, some species are highly sensitive to human impacts and others are more adaptable. From a management perspective, it is the declining group that is of primary concern to conservation practitioners.

As for ecosystem status, the best information available at the national scale is a series of maps produced by Global Forest Watch Canada, a now-defunct ENGO. These maps show the distribution of remaining intact landscapes in Canada as well as the distribution of various types of industrial activity (see Chapter 2). The federal government also conducts national-scale environmental monitoring, but its programs are motivated mainly by human health and socio-economic concerns. The emphasis is on attributes such as water pollutants, acid rain, pesticide use, greenhouse gas emissions, and forest restocking rates (SC 2011). These attributes are important, but measuring them provides little insight into the status of biodiversity at the ecosystem level.

Additional assessments of ecosystem status are provided by monitoring and research at the regional scale. For the most part, this information is collected to assess the impacts of specific land-use practices.

General Causes of Decline

We will discuss the causes of species declines in terms of agents and disturbance processes. To be clear, these disturbance processes do not constitute threats in all circumstances or to all species. Species vulnerability is determined by a combination of exposure to disturbance and sensitivity to disturbance. In this chapter, we will focus on the exposure aspect (what and where), leaving the discussion of species-level factors to Chapter 6.

Two types of disturbances affect wild species: natural disturbances, such as fire and drought, and **anthropogenic** (i.e., human origin) disturbances, such as agricultural land conversion and the release of industrial pollutants. Because species have been exposed to natural disturbances over evolutionary time, they have developed adaptations for coping with them. Populations may undergo short-term declines but normally have the resilience needed to bounce back. In contrast, many species are unable to accommodate the extent, intensity, and unique features of modern anthropogenic disturbances. Thus, anthropogenic disturbances account for almost all the long-term population declines that have occurred over the past century. Natural disturbances can be an important risk factor for species once long-term declines from anthropogenic impacts have reduced their populations to a small size (see Chapter 6).

Habitat alteration is the most important type of anthropogenic disturbance, affecting over 80% of Canada's species at risk (McCune et al. 2013). In some cases, habitat alteration takes the form of complete habitat loss, with nearly total replacement of natural ecosystem components with anthropogenic components (e.g., a wheat field). As habitat is lost from a region, the remaining habitat becomes increasingly **fragmented**, and this magnifies the impacts of the loss (Fahrig 2003).

For most species, fragmented patches of habitat cannot support the same populations that the equivalent amount of contiguous habitat can (Laurance 2008). In part, this is because habitat islands have more exposed **edge** than contiguous habitat (Fig. 5.2). For many species, edge constitutes poor quality habitat because it is subject to different micro-climates, different disturbance regimes, and different inter-specific interactions than interior habitat (Gehlhausen et al. 2000). Fragmentation also reduces connectivity and can interfere with territory establishment. All of these factors are species dependent because species perceive their environments at different scales and are sensitive to different factors.

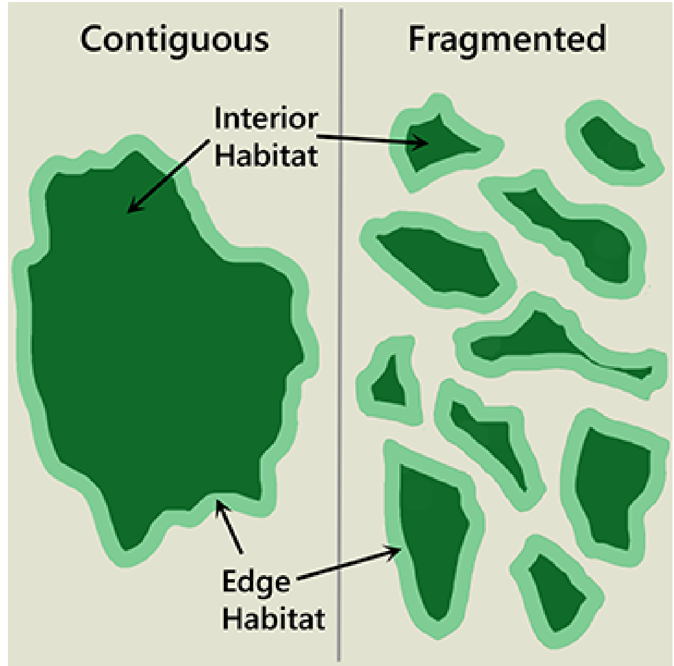


Fig. 5.2. When habitat becomes fragmented, the ratio of edge to interior habitat increases. In this illustration, the total area of habitat is the same in both panels but the amount of edge is much higher in the right panel.

The relative importance of habitat loss and habitat fragmentation continues to be debated. Some conservation scientists maintain that habitat loss is the overriding factor (Fahrig 2013), whereas others argue that fragmentation has an additive effect (Hanski 2015). As is often the case, both sides present valid arguments. There is good empirical evidence to suggest that habitat fragmentation has additive effects for some species in some circumstances (Ewers et al. 2007; Haddad et al. 2017). But it is also true that habitat loss and fragmentation are so closely correlated in most instances that they cannot be disentangled (Smith et al. 2009). Therefore, it is best to view these processes as two sides of the same coin. Moreover, we should recognize that the effects are nonlinear. The impacts of both processes are magnified when little habitat remains (Fahrig 2003).

Roads and other linear disturbances are a form of habitat loss that merits special mention (Fig. 5.3). In addition to habitat conversion, linear disturbances alter predator-prey dynamics, facilitate invasion by alien species, and cause soil erosion (Trombulak and Frissell 2000; Raiter et al. 2018). They also provide human access, which is problematic in its own right. For some species, like the marten, improved human access to remote areas exposes animals to increased harvest (Tigner et al. 2015). For others, like frogs, mortality from vehicles is most important (Eigenbrod et al. 2008). Some species, including grizzly bears and caribou, tend to avoid roads, which results in a functional loss of habitat (Dyer et al. 2002; Northrup et al. 2012). Once a species-specific threshold of linear disturbance density is reached, animal populations may exhibit reduced abundance and range contraction (Bayne et al. 2008; Tigner et al. 2015).

Fig. 5.3. A logging access road. Credit: S. Gunsch.

Aquatic systems are also threatened by road development. If an industrial access road happens to be built in the vicinity of a lake, access to the shoreline generally follows (Hunt and Lester 2009). With access, comes fishing and the depletion of preferred species. A study of lake trout in the boreal lakes in Ontario showed that abundance was reduced by half once access was developed (Kaufman et al. 2009). Roads also fragment aquatic habitat by blocking the natural flow of streams. Culverts are supposed to maintain this flow, but a significant portion fail over time due to poor design or installation (Fig. 5.4). An investigation in Alberta found that 50% of culverts were not functioning properly, suggesting that thousands of kilometres of streams were being fragmented (Park et al. 2008). Road construction is also a major source of erosion, leading to increased sediment in rivers (Kreutzweiser et al. 2013).

Fig. 5.4. A road culvert that has been poorly installed, creating a barrier to upstream movement of aquatic life. Credit: Sickter.

To the north of Canada’s agricultural region, human activities mostly result in reduced habitat quality rather than complete habitat loss. We will examine specific examples in the next section. Most entail ecosystem simplification, as reflected in species composition, ecological structures, and ecological functions. A decline in these attributes, relative to an unaltered system, is referred to as a loss of **ecological integrity**.

Other major contributors to species declines include (Venter et al. 2006):

- **Overharvesting.** This results from excessive hunting, fishing, trapping, and the removal of trees and plants for human use. In addition to declines in the abundance of the targeted species, there can be ripple effects throughout the biotic community from altered food web interactions.
- **Toxic chemicals.** These include industrial and residential pollutants as well as agricultural chemicals. These chemicals can affect the reproduction and survival of animals and plants, depending on their level of exposure and sensitivity. In the case of fertilizers, the issue is growth enhancement of certain species, resulting in ecological disruption.
- **Alien species.** The introduction of alien species can be inadvertent or deliberate. Alien species compete with and can displace native species, altering community structure and function.
- **Climate change.** Rising temperatures are changing the abundance and distribution of most species. The mechanisms and effects will be discussed in Chapter 9.

There are, of course, many other threats to species that are less extensive or are mainly a concern to species on the verge of extinction. Such threats include vegetation trampling from all-terrain vehicles, collisions and noise arising from ship traffic, bird mortality from windmill collisions, and the disturbance of nesting sites to name just a few.

The various forms of anthropogenic disturbance have all increased over time, coincident with technological advances and the growth of Canada's human population. Thus, population growth and technology can be considered indirect drivers of species declines.

Major Threats by Region

Having sketched out the main causes of species decline, we will now delve into the details. To do this, we need to drop down to the sub-national level because different parts of the country experience different patterns of anthropogenic disturbance. For our purposes, it is useful to define three broad terrestrial zones, which we will refer to throughout the text as the Agricultural South, the Industrial Forest, and the Far North (Fig. 5.5). Marine environments constitute a fourth zone. In the following sections, we will examine the major types of disturbance within each zone and describe their connection to species declines.

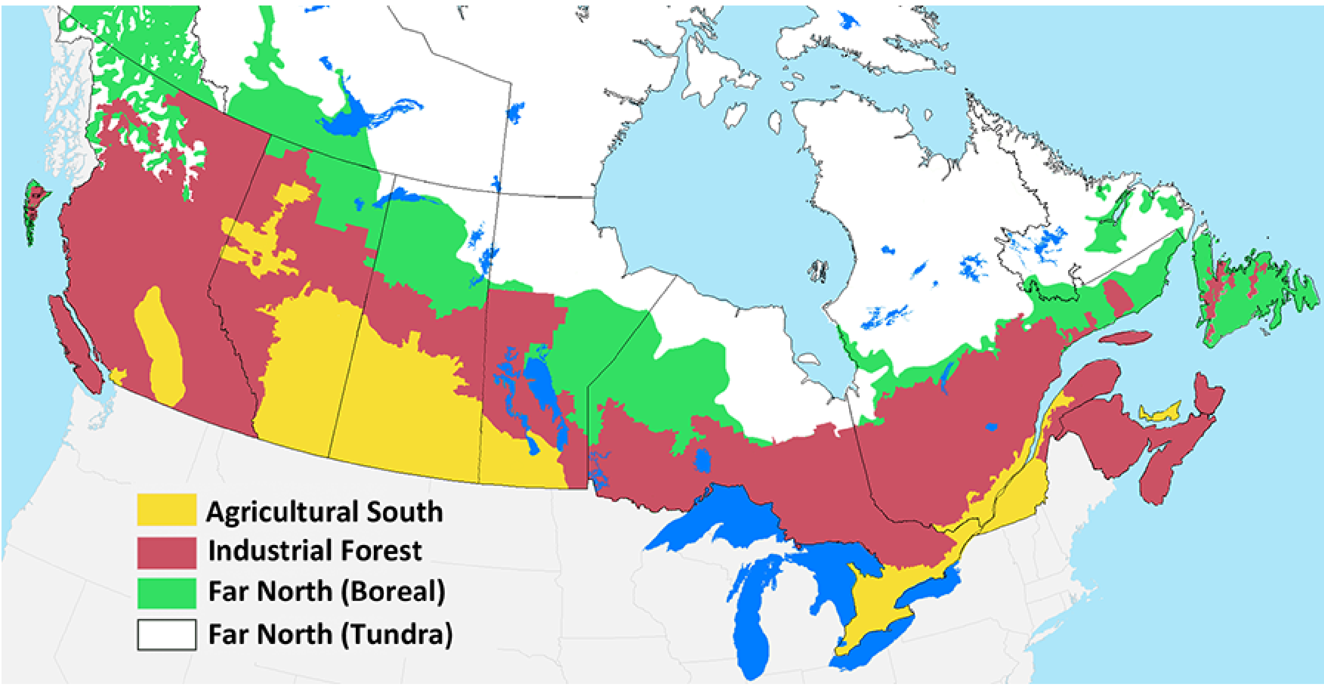


Fig. 5.5. Canadian regions classified by the major types of biodiversity threat present.

The Agricultural South

The Agricultural South zone (Fig. 5.5) includes both agricultural lands and most of Canada's population. This region has experienced the highest rate of biodiversity decline in Canada over the past century, and the majority of Canada's species at risk are located here (Kerr and Cihlar 2004; CESCC 2022). Not only does agriculture involve intensive land use, it has selectively targeted two specific ecosystem types: the Prairies Ecozone in the West and the Mixedwood Plains Ecozone in southern Ontario and Quebec (Fig. 2.5). No safe hinterland remains to provide habitat for the species in these two ecozones. Only 4.0% of the Prairies and 2.4% of the Mixedwood Plains exists within protected areas (ECCC 2022). Moreover, most of these protected areas are small and heavily biased to dry and unproductive rangelands (Deguise and Kerr 2006).

Habitat loss and degradation, through agricultural conversion and urban development, is the main threat to biodiversity in this region. Most of these losses occurred in the early twentieth century and, for the most part, have been permanent. Agricultural conversion is still occurring in some parts of the country, especially the interior of BC and the Peace River region of Alberta and northern BC (SC 2014b). Agricultural landscapes also feature an extremely high density of roads (Fig. 2.4). Natural disturbance regimes have been disrupted through the suppression of wildfires and the loss of native grazers (Campbell et al. 1994).

Declines in biodiversity are closely associated with the intensity of agricultural use, which has steadily increased over the years (Kerr and Cihlar 2004; Javorek and Grant 2011). As agricultural intensity increases, culminating in monoculture crop production, terrestrial ecosystems are progressively homogenized and simplified (Fig. 5.6). Wetlands in areas of intensive agriculture are also affected, through drainage, sedimentation, eutrophication, and salinization (Bartzen et al. 2010). Intensive agriculture also entails the widespread application of agricultural chemicals (Gibbs et al. 2009).

Fig. 5.6. An example of intensive agriculture. Collectively, soil tillage, wetland drainage, and the application of herbicides and pesticides leave little or no natural structure. Credit: Aqua Mechanical.

The simplified ecosystems of intensively managed agricultural lands can support only the most adaptable of species. Less adaptable species tend to concentrate in remnant patches of native habitat. Such lands account for approximately 30% of agricultural lands in the Prairie provinces, and 25% of agricultural lands in southern Ontario and Quebec (Javorek and Grant 2011). Most of the remnant patches are small and widely scattered, except in the rangelands found in the driest parts of Alberta and Saskatchewan. Many species lack the resilience needed to thrive in such fragmented and altered landscapes.

The threats to biodiversity from habitat degradation in the Agricultural South are exacerbated by agricultural chemicals and pollution, which affect both terrestrial and aquatic systems. Agricultural herbicides and pesticides can be directly toxic to wildlife species and can also reduce their food supply (Mineau et al. 2005). As a case in point, many species of native bees are currently in decline and these declines have been linked to pesticide exposure (Colla 2016). In the case of fertilizers, the problem is that a subset of species tend to benefit and become dominant, thereby decreasing overall species diversity (Haddad et al. 2000). Acid rain can cause declines in species with low acid tolerance, reducing species richness in regions with low buffering capacity (Vinebrooke et al. 2003).

Alien species present another threat to biodiversity in the Agricultural South. Most alien species are vascular plants, the bulk of which were inadvertently or deliberately introduced by immigrants from Western Europe. Today, 26% of known vascular plant species in Canada are non-native (Table 5.1). Of these, 486 are considered weedy or invasive (CFIA 2008). Many of the invasive species are agronomic grasses, such as smooth brome and crested wheatgrass, that have escaped from pastures and infiltrated native grasslands (McClay et al. 2004). Others are weeds, such as Canada thistle and leafy spurge. These alien species compete with and displace native plant species, altering community structure and function. This can have cascading effects throughout the system (Simberloff and Von Holle 1999). For example, the diversity of grassland birds is reduced where conversion to crested wheatgrass results in simpler habitat structure (Sutter and Brigham 1998). In addition, alien weeds are the cause of most herbicide use.

Fig. 5.7. A lake trout parasitized by sea lampreys, an alien species that entered the Great Lakes in the early 20th century. Credit: US Geological Survey.

Invasive alien species are also a major concern in aquatic systems (Dextrase and Mandrak 2006). A wide range of taxonomic groups are involved, including plants such as purple loosestrife, invertebrates like the zebra mussel, and various non-native fish species. Invasive alien species, in combination with overfishing and pollution, are responsible for the loss of much of the original biotic community of the Great Lakes (Mandrak and Cudmore 2010). Alien species entered the Great Lakes via many routes, including the Welland Canal, the ballast water of large ships, the undersides of fishing boats, and various forms of purposeful and inadvertent release by humans (Fig. 5.7). The 185 alien species that currently reside in the Great Lakes now dominate the system (CCRM 2010).

Most other aquatic systems are not as heavily altered as the Great Lakes, but invasive species are still a concern. The zebra mussel has now invaded Lake Winnipeg, the common carp is established in several provinces, the highly invasive Prussian carp has recently been detected in southern Alberta, and purple loosestrife is widespread (Elgin et al. 2014). The deliberate and often unauthorized introduction of non-native fish species for the purpose of sport fishing has also disrupted native fish assemblages in many lakes (Chapleau et al. 1997).

Alien species also include a variety of pests and disease agents, many of which target specific hosts. For example, the fungus responsible for white-nose syndrome affects only bats. After its introduction to North America, it caused the collapse of little brown bat populations in eastern regions and is now quickly spreading westward (Frick et al. 2010). Many prominent pests, such as the emerald ash borer, gypsy moth, and the Asian long-horned beetle are invasive insects that target trees. In the future, as global warming progresses, Canada's harsh climate will become less of a barrier for alien species, and we will see more of them enter.

Hunting, which was once a major threat to species in the agricultural zone, is much less of a concern today, though the legacy of past extirpations remains. Hunting of mammals and birds in this region is now well controlled and is mostly directed to species with large populations and high reproductive capacities, such as white-tailed deer and various waterfowl. In fact, in contrast to many other birds, waterfowl have shown a strong, increasing population trend since 1970 (NABCI 2019).

While hunting may no longer be a major concern in the Agricultural South, the same cannot be said of fishing. Recreational fishing regulations that focus on fish size and daily limits provide only a blunt instrument for managing total harvest because they do not control the total number of anglers using the resource. Fisheries involving rainbow trout in south-central BC, walleye and pike in Alberta, and lake trout in southern Ontario have all collapsed as a result of regulated recreational angling (Post et al. 2002). As a general rule, declines in recreational fish species occur in proportion to their proximity to population centres (Post et al. 2002).

The commercial harvest of freshwater fish in Canada is now considerably smaller than the recreational harvest (Post et al. 2016). However, commercial harvest was a major cause of fish declines in the past and continues to be problematic in some areas (ECO 2011). Many of the species that were once a staple, such as Atlantic salmon, lake sturgeon, and lake trout are gone from many lakes or have populations too small to support commercial harvest. As these species became depleted, the industry moved on to other species, including walleye, yellow perch, and lake whitefish. The status of these stocks is variable; some are stable, while others have collapsed (Sullivan 2003).

Fig. 5.8. All-terrain vehicles frequently cause ecological damage when used off designated trails. Credit: Toivo.

In addition to these widespread threats, there are threats with a regional or local impact. One of the most notable is disturbance from recreational activities, especially off-road vehicle use (Fig. 5.8). These types of disturbances increase with proximity to urban centres. They are a particular concern for at-risk plant species, the majority of which list recreation as a threat factor (McCune et al. 2013). There is also some mining and oil and gas development in the Agricultural South, but we will defer our discussion of these industries to the next section.

An important feature of all these threats is that they tend to overlap spatially. Species in the Agricultural South are rarely impacted by just one threat but by the cumulative impact of many factors operating in concert. Only the most adaptable species can thrive under such conditions.

The Industrial Forest

The Industrial Forest zone (Fig. 5.5) is defined by the spatial distribution of commercial forestry operations, which run in a broad band across the country (Fig. 2.9). Other notable forms of industrial development in this region include mining, oil and gas extraction, hydroelectric development, and peat extraction (Figs. 2.10 and 2.14).

Declines in biodiversity have been much lower in the Industrial Forest than in the Agricultural South, and there are fewer species at risk (even after accounting for lower species richness in the north; Kerr and Cihlar, 2004). It is mainly species with high sensitivity to anthropogenic disturbance that have undergone major declines, in contrast to the Agricultural South where only most adaptable species have remained stable. These sensitive species include habitat specialists, like the brown creeper (Poulin et al. 2008), and species that avoid roads and other industrial disturbances, such as woodland caribou (Dyer et al. 2002).

Species in the Industrial Forest zone have fared better than their agricultural counterparts mainly because forestry companies in most areas have long been required to reforest harvest blocks using native species. Moreover, less than half of Canada's forests are under industrial management, leaving a large reservoir of intact forest (though most of this is in northern areas with low productivity). Finally, the pace of forestry operations is relatively slow—less than 0.5% of the total forest area is harvested each year (CFS 2022). Consequently, most forests have only been harvested once, or are still awaiting their first harvest (Venier et al. 2014).

Although habitat changes in the Industrial Forest zone are not as severe as in the Agricultural South, they are still the main cause of declines of biodiversity in this region. Forest habitat is degraded through permanent deforestation, fragmentation, changes in forest patterns and composition, and the development of roads and other infrastructure.

Deforestation currently occurs at a low rate. Since 1990, conversion to other uses has averaged approximately 50,000 ha per year, out of more than 350 million ha of total forest area (CFS 2022). However, several specific forest types have experienced large losses in the past from which they have never recovered. These include the Carolinian forests of the Great Lakes Lowlands (Suffling et al. 2003), red and white pine forests in Eastern Canada (Venier et al. 2014), deciduous forests along the northern fringe of the prairie grasslands (Timoney 2003), and Douglas fir forests in coastal BC (CCRM 2010).

The deforestation that continues to occur today is primarily from agricultural expansion and industrial development (Table 5.2). Losses to agriculture occur mainly along the northern boundary of the agricultural zone (Hobson et al. 2002). Most of the industrial impact is from oil and gas exploration and development, which is centred in northern Alberta and northeast BC (Fig. 2.10). Mining is another contributor and it is widely distributed across the country (Fig. 2.14). Sporadic, localized forest losses also occur from flooding associated with hydroelectric development. Peat mining is not included in deforestation tallies, but it constitutes another form of habitat loss in this zone (Kreutzweiser et al. 2013). Most peat mining occurs along the southern fringe of the boreal forest.

Table 5.2. The annual rate of permanent deforestation in Canada, by sector, for 2020.[1]

Sector	Area (ha)
Agriculture	22,378
Oil and gas + mining	15,144
Infrastructure	9,637
Hydroelectric	1,101
Forestry	1,092
Total	49,352

[1]Source: CFS 2022.

Though the rate of deforestation is low at the national scale, it can be significant at the regional scale. In some parts of northern Alberta, the annual area of forest clearing by the oil and gas sector rivals that of the forest industry (Schneider et al. 2003). Furthermore, there has been limited progress in restoring native vegetation to wells and mines after production has ceased (Osko and Glasgow 2010). Such sites have typically been restored to grass rather than forest. Consequently, forest clearing for oil and gas development and mining tends to be semi-permanent, resulting in impacts that accumulate over time (Nitschke 2008; Pickell et al. 2015).

The annual amount of permanent deforestation is dwarfed by the 700,000–800,000 ha harvested each year by the forest industry (CFS 2022). Forest harvesting is considered a transient disturbance, not deforestation, because harvest blocks must be regenerated. Nevertheless, it has widespread effects on forest structures and patterns (Grondin et al. 2018). These changes arise from the differences that exist between conventional forest harvesting and natural forms of disturbance, such as wildfire and insect attacks.

The most important difference between harvesting and natural disturbance relates to the age structure of the forest (Cyr et al. 2009). Natural disturbances are generally random, so some stands remain undisturbed for long periods simply by chance. This results in an age distribution featuring a long "tail" of older stands (Fig. 5.9). Through successional processes, the undisturbed stands develop unique structural and compositional attributes as they mature, which help to support a variety of specialist species (see Chapter 7).

In contrast, conventional forestry operations specifically target older stands because of their high wood volume and value. Moreover, fast-tracking the harvest of old stands can avert losses from fire, insect outbreak, and natural mortality, which occur at a higher rate in older age classes. Over time, the selective harvest of older stands truncates the forest age structure at the preferred harvest age (Fig. 5.9; Bergeron and Fenton 2012). This in turn causes declines in old-forest specialist species (Schmiegelow and Monkkonen 2002).

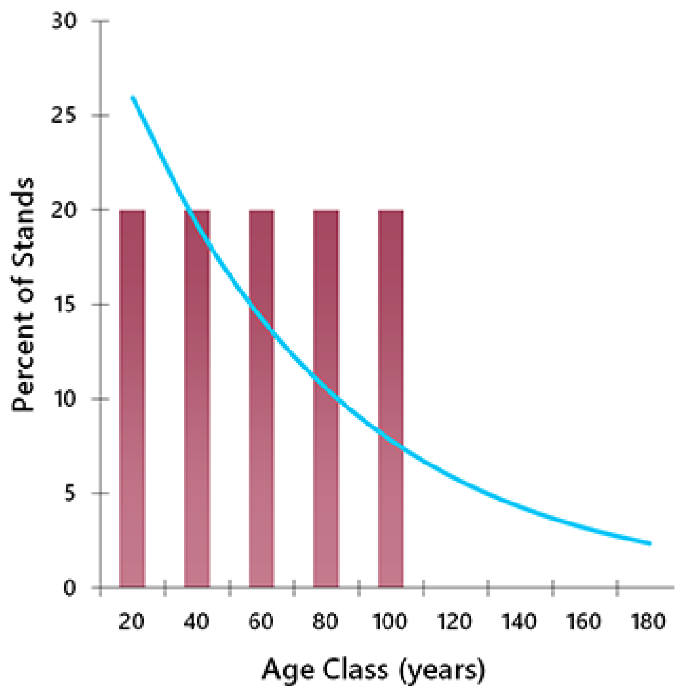


Fig. 5.9. Conventional forest harvesting targets older stands and leads to a truncated age structure. In contrast, the random nature of natural disturbances permits some stands to escape disturbance for long periods. This example compares harvesting at 100 years (columns) to a fire regime that randomly burns 1.5% of the landscape each year (blue line).

The ecological effects of age-class truncation are readily observed in forests with a long history of harvesting, such as the boreal forests of Finland. In Finland, only 6.5% of stands on managed landscapes are over 140 years old, whereas 50% of stands within protected areas are in this age group (Virkkala and Rajasarkka 2007). As might be expected, most of Finland's threatened bird species are old-forest specialists, and they are now found mainly in protected areas. Forest age structures in Canada are not yet as skewed as those in Finland, but only because the bulk of our commercial forests were only brought into production in the last half of the twentieth century. The full story of forestry impacts in Canada has yet to be revealed.

Another type of habitat at risk from forestry is burned forest (Hannon and Drapeau 2005). In managed landscapes, salvage logging removes trees that have been burned, and fire suppression reduces the overall amount of burning. This again presents a problem for habitat specialists. In natural systems, burned stands are targeted by species like the black-backed woodpecker that exploit insects which attack dead trees.

Forest harvesting also tends to simplify forest structures and patterns (Doyon et al. 2008). Whereas fires leave standing dead trees, patches of live trees, and coarse debris, all of which contribute to habitat complexity, conventional clearcutting leaves little behind. Furthermore, in the mixed coniferous and deciduous forests of the eastern and western boreal region, silvicultural practices tend to produce pure coniferous or deciduous stands, rather than mixed stands, reducing niche complexity and species richness (Hobson and Bayne 2000; Boucher et al. 2009). Finally, conventional clearcutting results in simple spatial patterns, rather than the complex shapes and variable patch sizes produced by fire (Fig. 5.10). In all these cases, it is not the individual changes that are important, but the cumulative transformation that occurs across large areas. Simplified forests have more generalist species and fewer specialists than forests with complex niche structure (Hobson and Bayne 2000; Zhang et al. 2013).

Fig. 5.10. An aerial view of the "checkerboard" pattern that results from conventional two-pass forest harvesting. Credit: Google Images.

Another threat to wildlife in the Industrial Forest zone is the vast network of roads, pipelines, and power lines that permeate the region, providing access to forestry harvest blocks, oil wells, and mines as well as routes for moving products to market (Fig. 2.4). In contrast to harvest blocks, which are reforested, roads and utility corridors are usually permanent. Moreover, new roads and corridors continue to be built each year, as remote areas are accessed for the first time.

Seismic lines, used for oil and gas exploration, add to the density of linear disturbances in forested landscapes (Fig. 5.11). In the 1980s and 1990s, an average of 46,000 km of seismic lines were approved for the forested area of Alberta each year (AEP 1998). Seismic exploration has also been intensive in northeast BC. These older lines, usually 6–8 m wide, remain as legacy disturbances today.

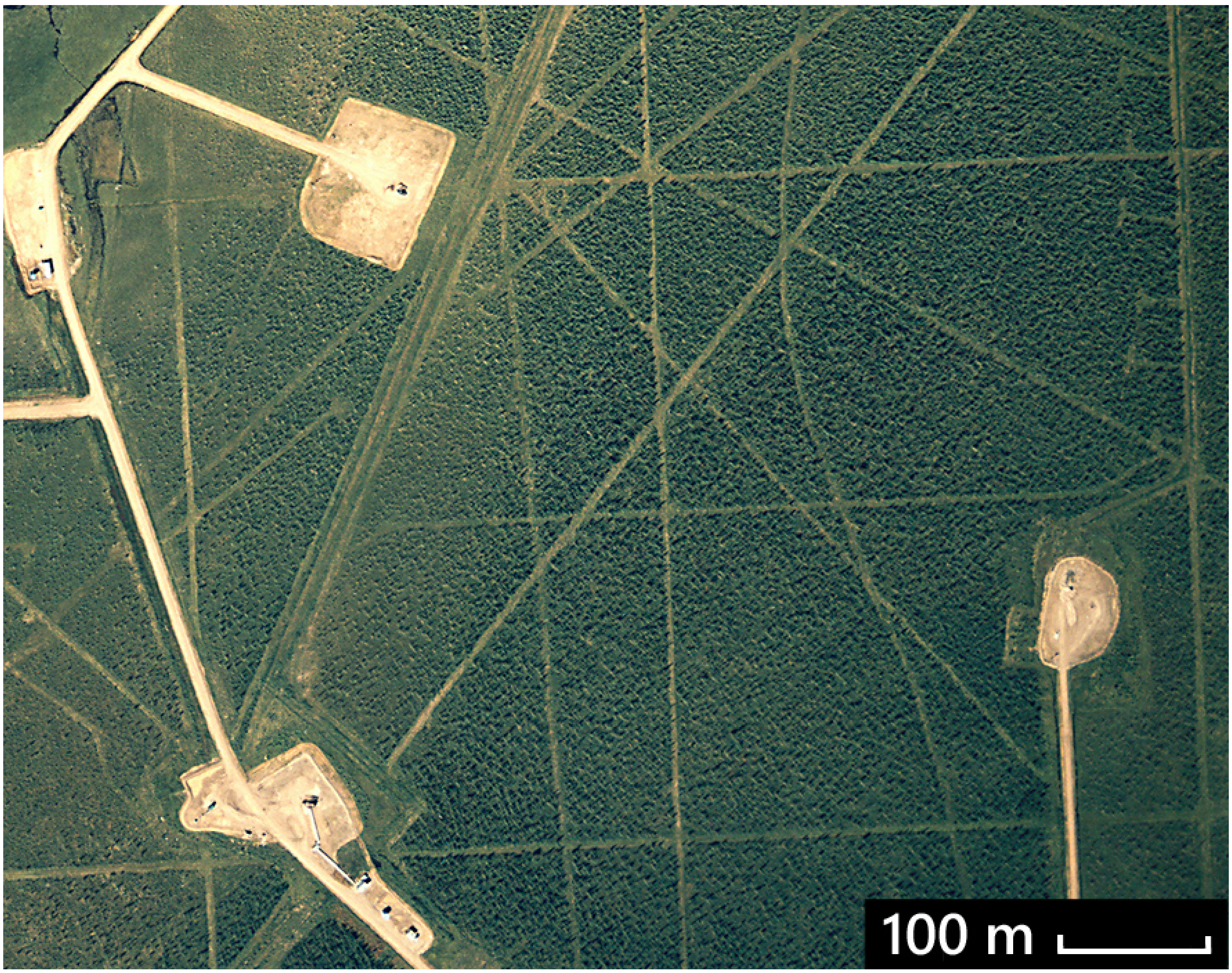


Fig. 5.11. The industrial footprint of oil and gas development in northern Alberta, illustrating well sites (pale squares), access roads (thick lines), and seismic lines (thin lines). Credit: Air Photo Services.

Exploration companies were not required to reforest these lines, and natural regeneration has been slow, often because of ongoing use by all-terrain vehicles and snowmobiles. A study of lines created in the 1960s and 1970s found that 65% remained in a cleared state 35 years after they were created, and only 8% were fully reforested

(Lee and Boutin 2006). Technology now exists to create lines that are only 2.5 m wide, but it is not used consistently.

Biodiversity in the Industrial Forest is also affected by pollution, particularly from mining. Ore bodies often contain toxic elements, including cadmium, lead, and arsenic. Once the rock is ground into fine-grained tailings, these elements can migrate into the environment if not properly stored. Other types of tailings produce acid when they come in contact with oxygen and water, promoting the leaching of metals and the acidification of ground and surface waters (Shang et al. 1999). The risk of release into the environment is greatest for abandoned mines, of which there are thousands (MacKasey 2000). Accidents also happen in active mines, as demonstrated by the Mount Polley tailing pond failure in BC in 2014, which released millions of cubic metres of toxic slurry into the adjacent water system (IEEIRP 2015).

Pollution also arises from other industries. Oil and gas production causes soil and water contamination through pipeline ruptures, accidental spills, drilling discharges, and improper waste disposal. In the oil sands, toxic tailing ponds cause mortality in waterfowl and other bird species that land on them inadvertently. In the forestry sector, pulp mills release organochlorines and other processing byproducts into local waterways.

Air pollution is another common byproduct of industrial processing and can affect wide regions. One of the main threats to biodiversity is the acidification of lakes and forests through acid rain, particularly in the Shield, where little natural buffering capacity exists (Schindler 2001). The sources of SO_2 and NO_2 that create acid rain include coal-fired generators, industrial emissions, and vehicle emissions.

Lastly, biodiversity in the Industrial Forest is threatened by alien species. Insect pests and disease organisms, such as Dutch elm disease, white pine blister rust, and gypsy moth are of particular concern (Allen and Humble 2002). Alien plants are also a threat, especially in disturbed sites and riparian areas (Rose and Hermanutz 2004). The spread of many alien species, such as earthworms (Cameron and Bayne 2009) and agronomic grasses (Sumners and Archibold 2007), is facilitated by roads. Some of these alien species act as sporadic disturbance agents, whereas others integrate into the forest ecosystem, changing competitive relationships and altering ecological processes.

The Far North

The Far North is the largest zone, encompassing all lands north of the Industrial Forest zone (Fig. 5.5). It is not entirely pristine, but the vast majority is roadless and undisturbed (Fig. 2.4). The roads that do exist are mostly used to provide access to scattered mining developments. Some access has also been developed in association with oil and gas development, mostly in the Yukon and along the Mackenzie River in the NWT.

Since most of this region remains undisturbed, the threats to biodiversity are low relative to the southern regions. However, because of the harsh conditions in the north, resilience to disturbance is also low, and vegetation regrowth is very slow (Mallory et al. 2006). Consequently, when disturbance does occur, it has a greater effect on biodiversity than in the south.

As in the south, it is not just the footprint of individual mines and oil wells that needs to be considered, but also the cumulative effect of development at the regional scale, including the expansion of access, provision of housing for workers, and creation of other infrastructure. These cumulative effects can disrupt animal movement patterns and reduce habitat quality for species sensitive to disturbance, including caribou, grizzly bears, wolves, and wolverines (Johnson et al. 2005).

The other main threat to biodiversity in the Far North is hunting. Although Indigenous communities in the north have a long tradition of sustainable harvesting, human populations are higher today than in the past, and snowmobiles and rifles have dramatically increased the efficiency of hunting. The declines of some species, such as caribou, have been attributed to unsustainable harvest rates in combination with other factors (Boulanger et al. 2011). This is currently a point of management controversy in the Far North (Nesbitt and Adamczewski 2013; Parlee et al. 2018). Other flash points include the trophy hunting of polar bears and the traditional hunting of endangered whales (Freeman and Wenzel 2006).

Because of the remoteness and lack of access, the status of species in the Far North is difficult to determine (NABCI 2019). Most monitoring efforts have been directed to species that are socially important and relatively amenable to measurement, such as caribou and polar bears. For most other species, the impacts of human activities are not well known.

Marine

The most important threat to biodiversity in marine environments is fishing (Baum and Fuller 2016). The economic value of commercial fishing has led to conflicts between protection and exploitation, and management outcomes have generally been poor (McDevitt-Irwin et al. 2015). The collapse of the northern cod fishery is the most notorious example, but it is just one of many fisheries that have collapsed. Of 125 commercially harvested fish and invertebrate stocks with recent status assessments, only 24% are healthy (Baum and Fuller 2016). Forty marine species have been assessed by COSEWIC as endangered or threatened, but most have been denied listing under SARA on socio-economic grounds (McDevitt-Irwin et al. 2015).

Fishing has several distinct effects on marine species. Direct mortality from harvesting is the most obvious and can result in the collapse of targeted species if the rate of harvest is too high. The collapse of targeted fish stocks can in turn affect other species by disrupting the normal food chain (Fauchald 2010). Commercial fishing also causes collateral damage through the inadvertent harvest of non-target species (bycatch). In the Pacific groundfish trawl fishery, discarded bycatch amounted to 20% of the total biomass caught between 1996 and 2006 (Driscoll et al. 2009). Finally, fishing methods based on bottom trawling can destroy seafloor habitats. This has been identified as a major threat to cold water corals and sponges, which are found off Canada's eastern and western coasts (Wallace et al. 2015).

Another threat to species in marine environments is pollution. Most industrial and residential liquid waste eventually makes its way into rivers that discharge into the ocean. Thus, oceans become contaminated with myriad chemical products. These chemicals are rapidly diluted, to be sure. Even so, some enter the food chain and undergo **bioamplification**. In higher-level predators, like killer whales, exposure can reach toxic levels (Alava et al. 2016). Ocean contamination with plastics is also a growing concern, though the population-level effects are not yet understood (Rochman et al. 2016). Finally, catastrophic pollution occasionally happens through oil spills from tankers and offshore drilling installations (Fig. 5.12). This has been one of the main points of resistance to the construction of the Trans Mountain pipeline from Alberta to the BC coast.

Fig. 5.12. Oil-covered rocks in Prince William Sound after the Exxon Valdez spill. Credit: Alaska Resources Library & Information Services.

For some species, particularly whales, collisions with ships and entanglement in fishing gear are significant threats (Williams and O'Hara 2010). Incidence is highest along the southern coast of BC and in the Gulf of St. Lawrence, where whales must contend with high levels of maritime traffic. For the North Atlantic right whales, the issue has become critical, with 12 whales killed in the Gulf of St. Lawrence 2017. Most died from blunt trauma from ship strikes, though at least two died from entanglement in fishing gear (Daoust et al. 2018).

CHAPTER VI

SPECIES-LEVEL CONSERVATION

Species-Level Conservation

With this chapter, we transition to applied conservation. We will begin with conservation approaches tailored to the needs of individual species, often referred to as "**fine-filter**" methods. Because of knowledge and capacity constraints, only a small subset of species, referred to as "**focal species**," receive such individualized attention (Box 6.1). Conservation approaches designed to support biodiversity in general are referred to as "**coarse-filter**" methods and they are the subject of Chapter 7.

Box 6.1. The Coarse Filter, Fine Filter, and Focal Species

The coarse-filter/fine-filter approach to conservation was initially developed in the context of protected area planning (Hunter et al. 1988). The coarse-filter approach entailed protecting a representative sample of broad ecosystem types, thereby supporting the habitat needs of most species. The fine-filter approach was intended to protect species that slipped through the metaphorical coarse filter because of unique habitat requirements. Today, these terms are used more loosely, often simply indicating the scale of conservation effort: individual species versus overall biodiversity (Tingley et al. 2014). Strictly speaking, it is best to refer to the targets of species-level conservation as focal species because these species are not limited to those "missed" by the coarse filter.

Focal species are selected because of their ecological or social importance, and they come in a variety of overlapping forms (Lambeck 1997; Noss 2003):

- **Species at risk.** These are species that have been formally designated as threatened, endangered, or at risk under federal or provincial legislation. The Vancouver Island marmot and eastern wolverine are examples.

- **Keystone species.** Keystone species have a disproportionately large effect on their environment relative to their abundance. They play an essential role in determining the structure, function or productivity of an ecosystem. Beavers and prairie dogs are examples.
- **Sensitive species.** These are species that are uniquely sensitive to habitat disturbance and are thus dependent on habitat protection. Habitat specialists like the black-backed woodpecker and brown creeper are in this category.
- **Flagship species.** These are charismatic species with popular appeal that serve as symbols and rallying points for stimulating conservation awareness and action. Woodland caribou and the giant panda are examples.
- **Game species.** Species that are hunted, fished, or trapped are often included as focal species because of their utilitarian value. The mallard duck and moose are examples.
- **Umbrella species.** Umbrella species have large area requirements and therefore use many different habitat types. They are selected as focal species because of their ability to represent the habitat needs of many other species. Grizzly bears and wolves are examples.

There are three main forms of focal species conservation. The oldest form involves the management of species that are harvested, and it places an emphasis on utilitarian values. The focus is on harvest management, with the aim of maintaining sustainable populations while maximizing the flow of benefits (Passelac-Ross 2006). Responsibility for this form of conservation rests mainly with provincial fish and wildlife departments and forestry departments, working under provincial legislation. The federal government has a lead role in the management of migratory birds, fisheries, and wildlife on lands under federal jurisdiction.

The second form of focal species conservation is concerned with the identification and management of species at risk. Responsibility for management is shared between the federal and provincial governments under the federal *Species at Risk Act* (SARA) and related provincial legislation. The emphasis is on species recovery, with the objective of achieving self-sustaining populations.

Lastly, conservation of focal species occurs in the context of resource management and land-use planning, usually in conjunction with generic coarse-filter measures. Responsibility for this form of management is shared between provincial governments and land users (particularly industrial operators), as directed under provincial biodiversity policies. The objectives vary by species and by management area, but the general aim is to maintain natural levels of abundance, to the extent possible given competing socio-economic objectives.

We will begin this chapter by reviewing the ecological theory that underpins species-level conservation and examining how field data and models are used to support applied conservation. We will then consider the practical aspects of species management, including both planning and "hands-on" activities. Our focus will be on approaches tailored to individual species, rather than broad landscape-level approaches, which are the subject of Chapter 7. We will conclude the chapter with an examination of the trade-offs inherent in species-level conservation and the approaches available for dealing with these trade-offs.

This chapter, through to Chapter 8, describes conservation as it is generally practiced today. Once this foundation is in place, we will turn, in Chapter 9, to the refinements to conventional practices needed to accommodate climate change.

Theoretical Foundation

Species-level conservation is fundamentally concerned with the processes governing **population dynamics** and how they are affected by external factors. Knowledge of these processes is gained through observational studies and demographic modelling. Models are tools that:

- Provide an organizing framework for the observational data we collect
- Allow us to explore the dynamics of the system, helping us to understand it more fully and to extract important principles
- Help us identify which parameters are most influential and where key uncertainties lie
- Allow us to make predictions about how the system will respond to alternative management scenarios

Some models are strategic in nature, sacrificing detail for generality. By incorporating relatively simple mechanisms that do not consider the details of any one system, they aim to capture the essential behaviour of many systems (Yodzis 1989). In this section, we will explore the theoretical insights fundamental to species-level conservation provided by strategic modelling. We will begin with the simplest of systems and then progressively add detail. In a later section, we will turn to tactical models, which capture the details of specific systems and are used as tools to support applied conservation.

Box 6.2. Species and Populations

For biologists, a species is a group of organisms capable of interbreeding under natural conditions. However, under SARA, "wildlife species means a species, subspecies, variety or geographically or genetically distinct population of animal, plant or other organism, other than a bacterium or virus, that is wild by nature" (GOC 2002, Sec. 2.1).

Much of the research and management that occurs under the heading of species conservation actually targets populations. A population is a group of organisms of the same species that live in a particular geographical area and which normally breed with one another. In practice, the level of interbreeding is usually unknown, and the term "population" is simply applied to local assemblages of individuals of the same species.

Insights from Simple Population Models

Every species has an intrinsic reproductive capacity, determined by the age at first reproduction, number of offspring per reproductive cycle, number of cycles per year, and so forth. Similarly, each species has an intrinsic rate of senescence that determines the maximum lifespan of individuals. Together, these two variables define the maximum population growth rate of a species.

In real-world settings, the maximum rate of population growth is rarely observed because reproduction and mortality are affected by a wide range of limiting factors. Common examples include competition for resources, consumption/predation, disease, and environmental disturbances. Some factors have a proportionately greater effect as population density increases. For example, competition for resources may be minor concern when a population is small, but a major limiting factor if its density becomes high. These are called **density-dependent** factors, and they serve as negative feedback mechanisms. Other factors, such as fire, have roughly the same proportional effect on populations, regardless of population density. These are called **density-independent** factors (Hayes et al. 1996).

A simplified illustration of density-dependent relationships is shown in Fig. 6.1. The growth rate of the entire population (total births minus total deaths) relative to density is provided in Fig. 6.2. The exact shape of these curves will vary from species to species and among regions because they depend on intrinsic species traits and the nature of the limiting factors in a given area. However, the general features of density-dependent population growth are widely applicable.

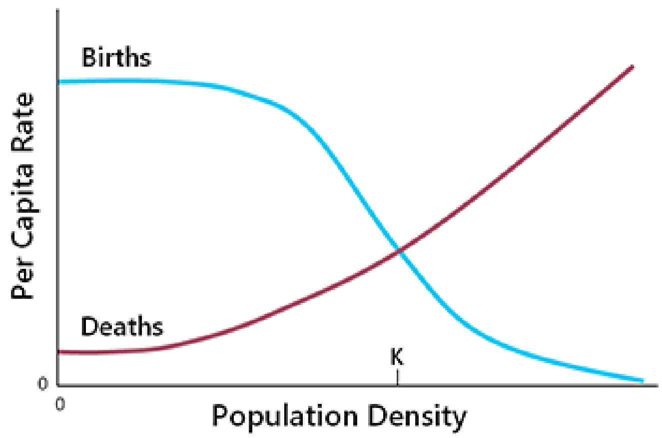


Fig. 6.1. Per capita birth rate and death rate as a function of population density, illustrating simple density-dependent relationships.

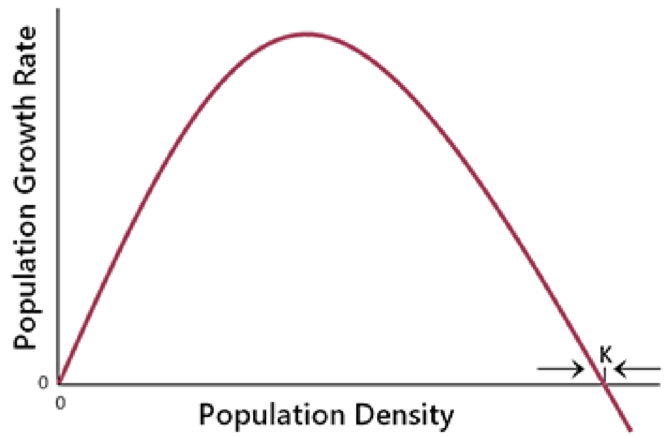


Fig. 6.2. Population growth rate as a function of population density, given the functional relationships shown in Fig. 6.1.

An important feature of these curves is that there exists a point, labelled K in Fig. 6.1, where the population is in equilibrium because the rate of births equals the rate of deaths. The point K is usually referred to as the **carrying capacity** (Yodzis 1989). Because of density-dependent feedback processes, the population will intrinsically revert back to K if it is perturbed (see Fig. 6.2). As such, K is an important point of reference for management.

The Natural Range of Variability

In our simple example, population size becomes static once the density equals the carrying capacity. But real-world populations tend to fluctuate in size because many of the factors that influence reproduction and mortality are intrinsically **stochastic** (i.e., exhibit randomness). Sporadic disturbances and environmental processes with inherent variability, such as weather, have the greatest effect (Lande 1993).

The time it takes a population to return to its equilibrium state after a perturbation is a measure of its **resilience** (Gunderson 2000). This rate is determined by the population growth functions we discussed earlier (Fig. 6.2). Under natural conditions, a population's resilience is generally sufficient to accommodate the disturbances it encounters, and so it fluctuates about its equilibrium value (Fig. 6.3). Species unable to accommodate such perturbations are eliminated through natural selection.

Given the stochasticity inherent in population dynamics, it is difficult to determine carrying capacity through direct observation. The **natural range of variability** (NRV) is often used instead as the reference state for management (Fig. 6.3). In areas where natural conditions prevail, the NRV of a given population can be determined through long-term monitoring. For populations in developed areas, NRV can be derived from historical data, extrapolated from nearby natural areas, or estimated using population models. Because of data limitations, only a rough estimate may be possible in such cases.

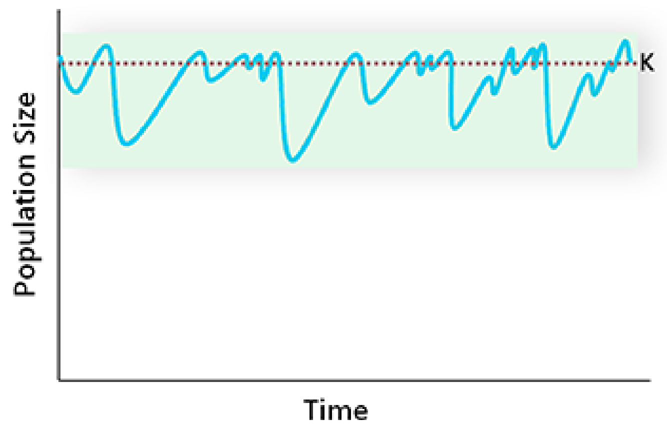


Fig. 6.3. Natural populations (blue line) fluctuate about their equilibrium level K (dotted red line) as a result of periodic environmental disturbances. The shaded area corresponds to the NRV.

Population Decline

Having explored the dynamic processes characteristic of natural systems, which feature fluctuations about an equilibrium point, we now turn to the processes involved in systematic population decline (Caughley 1994). A widespread cause of population decline is habitat loss. Individuals that are displaced through habitat loss cannot simply crowd into the remaining habitat because it cannot support a density greater than the carrying capacity, at least not for long. Therefore, losses in habitat area translate into direct losses in overall population size.

Population decline can also occur because of changes in per capita birth and death rates (Fig. 6.4). The list of proximate causes includes overharvesting, reduction in habitat quality, competition from alien species, pollution, and various other factors (detailed in Chapter 5). To understand the population dynamic implications, it does not matter too much whether birth rates decline or deaths rates increase (or both). The main consequences are a reduction in the intrinsic population growth rate and a lower carrying capacity (K_2 in Fig. 6.4).

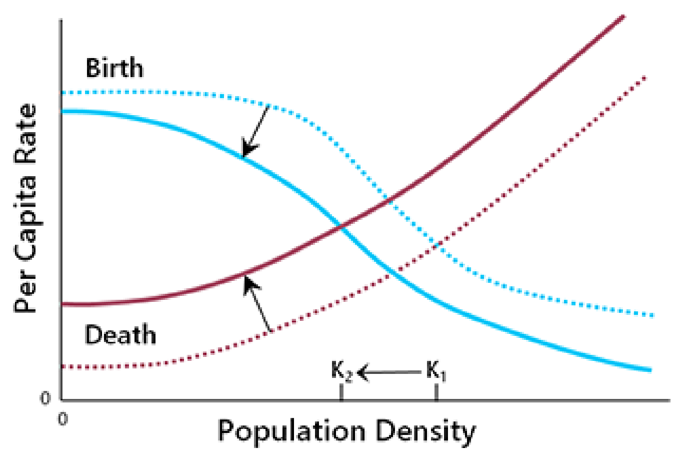


Fig. 6.4. An illustration of how the birth and death rate functions from Fig. 6.1 (dotted lines) might change as a result of anthropogenic disturbance (solid lines).

A reduction in the population growth rate can result in a range of outcomes (Hayes et al. 1996). In the extreme case, where mortality exceeds reproduction at all densities (implying a negative growth rate), the population will invariably go extinct. Large initial population size may slow the process, but will not prevent it, as illustrated by the extirpation of Canada's plains bison from overhunting.

If the intrinsic growth rate remains at least somewhat positive, the long-term outcome depends on the balance between the rate of population growth and the rate of environmental disturbances. Given sufficient growth potential, a population can recover from disturbances and remain within NRV. However, beyond a certain point, the rate of recovery may become too slow to keep pace with disturbances, resulting in a declining trend (Fig. 6.5).

Changes in the birth rate and death rate can also lead to a reduction in carrying capacity (Fig. 6. 4). If this happens, the population will equilibrate at a lower density, assuming that the growth rate is sufficient to maintain stability (Fig. 6.6). The danger here is that, if the new equilibrium density is very low, extinction may result from the demographic challenges of small populations (discussed below).

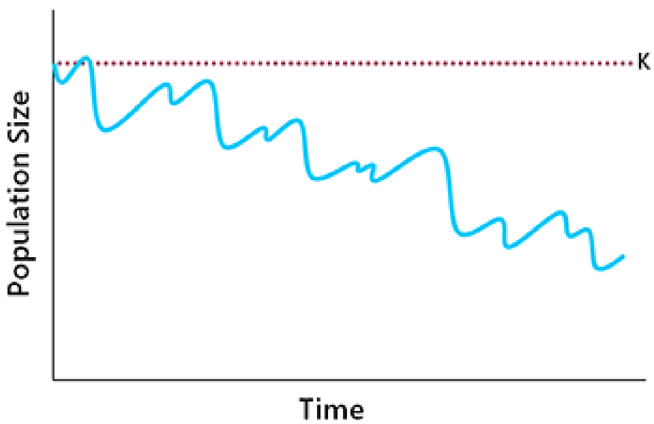


Fig. 6.5. If a population's intrinsic growth rate declines, it may be unable to recover quickly enough from periodic environmental disturbances to prevent a declining trend.

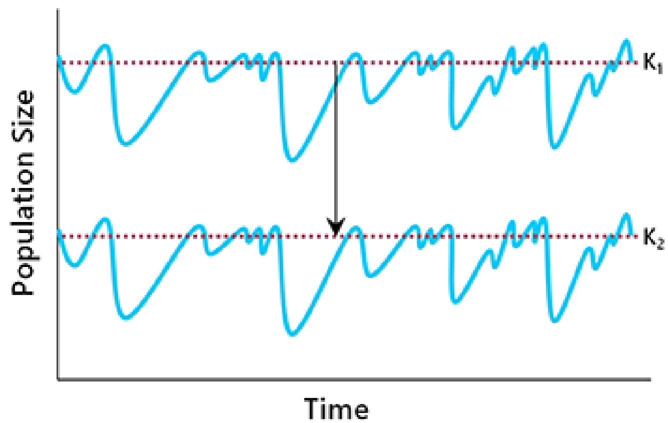


Fig. 6.6. When K is reduced, the population will equilibrate at a lower density, potentially exposing it to the demographic challenges of small populations.

In some cases, declining trends can be reversed through remediation of the inciting causes, allowing populations to re-establish their original NRV. For example, by banning DDT, hunting, and egg collecting, peregrine falcon growth rates have been substantially restored, and populations are now recovering (albeit, with initial help from captive breeding programs; COSEWIC 2007).

More commonly, mitigation efforts are constrained by socio-economic trade-offs that preclude full recovery (Traill et al. 2010). For example, plains bison have sufficient growth potential to return to their original NRV. But such recovery would require the naturalization of vast landscapes now used for agriculture, which is politically infeasible. We will explore these sorts of trade-offs later in the chapter.

Ecological Thresholds

In some cases, a species may exhibit a linear response to a given environmental driver (Fig. 6.7). For example, population size may decrease in direct proportion to the amount of habitat loss. More commonly, a species may show little response to low levels of the driver and then, at a certain point, exhibit a disproportionately large response. This transition point is referred to as an ecological threshold, and it results from nonlinear system dynamics (Kelly et al. 2015).

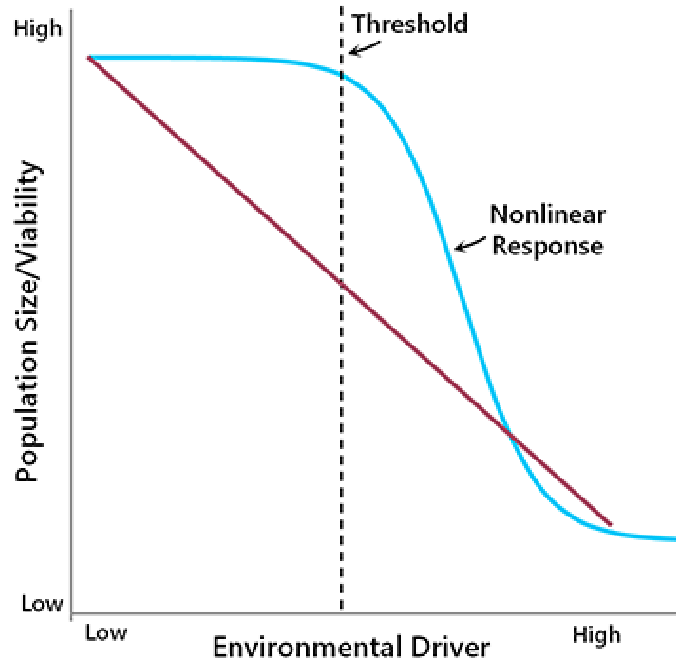


Fig. 6.7. Hypothetical response curves to an environmental driver illustrating linear and non-linear relationships. The ecological threshold, applicable to the nonlinear curve, is also shown.

An ecological threshold is typically observed when compensatory processes initially buffer the effect of an environmental driver. The driver's ecological effects become apparent once this buffering capacity is exceeded. For example, given a population at carrying capacity, harvesting at low levels may simply offset other forms of density-dependent control, whereas harvesting at higher levels may exceed the capacity for compensation and result in population decline (Boyce et al. 1999).

Though ecological thresholds are undoubtedly common in natural systems, they are difficult to quantify. Data must be collected across a broad range of disturbance levels, which is not always possible. Interactions with other drivers and ecological inertia can also complicate matters (van der Hoek et al. 2015).

When an ecological threshold can be reliably delineated, it facilitates the selection of management targets. In cases where ecological responses have not been well described, a linear relationship is usually assumed. In some cases, an ecological system may undergo qualitative changes once a species declines below a certain point, making recovery much more difficult (Gardmark et al. 2015). This is another form of ecological threshold. Northern cod provide an example. Intensive harvesting of cod released herring from predator control, leading to an altered ecosystem (Fauchald 2010). In this new state, an abnormally large population of herring suppressed cod recruitment through predation on cod eggs and larvae. Consequently, the path to cod recovery has been prolonged, despite the moratorium on fishing implemented in 1992 (Fig. 6.8).

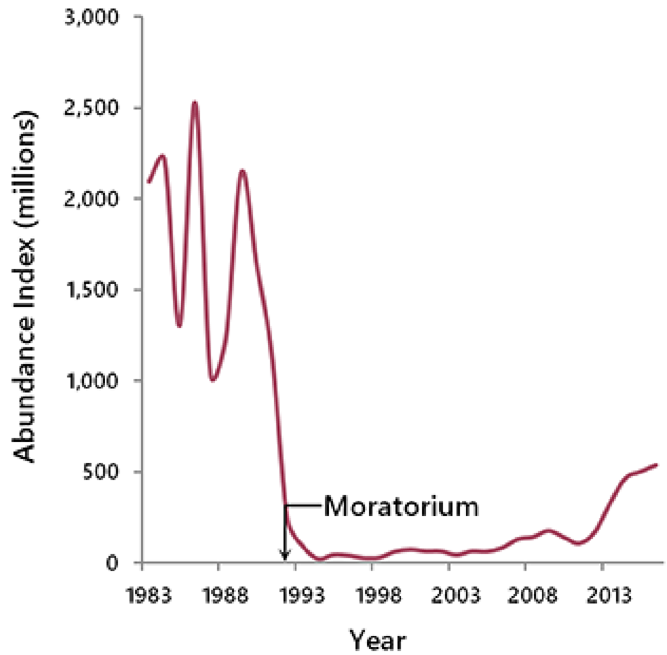


Fig. 6.8. The abundance of northern cod, 1983–2016, from autumn surveys. Stock levels remained collapsed for 20 years despite a moratorium on commercial harvesting implemented in 1992, but are now beginning to recover. Source: FOC 2017b.

Box 6.3. Traits that Predispose Species to Extinction

Several species traits are associated with an increased risk of extinction, either because they increase exposure to anthropogenic threats, or because they hinder the ability of a species to cope with these threats (Flather et al. 2011). It is worth noting that large mammals exhibit many of these traits, partially explaining their prominence as focal species (Carroll et al. 2003). Traits predisposing to extinction include:

- Reproductive strategies designed for stable environments, including late maturation, low reproductive potential, and low natural density
- Large body size and large home range size
- Habitat or diet specialization, particularly if associated with a restricted range
- Commercial value (e.g., for hunting or fishing)
- High sensitivity to common anthropogenic effects (e.g., vulnerability to pesticides)
- Long-range migration
- Colonial nesting

Extinction Dynamics

Once populations become small, the extinction risks from stochastic processes are magnified, and genetic and demographic factors also become important (Caughley 1994). The population threshold at which these processes become a serious concern varies by species and individual circumstances. However, we can use generic estimates derived from theoretical analyses to convey the relative importance of the mechanisms involved.

Environmental stochasticity is the most significant factor involved in extinction dynamics because it includes catastrophic events, like large fires, that can cause the direct mortality of large numbers of individuals (Lande 1993). Small populations face the highest risk because even relatively small disruptions can result in extinction (compare the two populations in Fig. 6.6). As a ballpark estimate, achieving long-term persistence in the face of environmental stochasticity may require a population of several thousand individuals (Soule 1987; Traill et al. 2010).

Genetic effects contribute to extinction risk in two main ways. Once populations become small (less than ~1,000 individuals) any further loss of individuals may result in the loss of **genetic variability** (Shaffer 1981; Lande and Barrowclough 1987). This reduces the capacity of individuals and populations to adapt to changing conditions. At even lower population sizes (i.e., less than approximately 50 reproductive individuals) the fitness of individuals may be further reduced through **inbreeding depression** (Saccheri et al. 1998). Inbreeding depression involves the increased expression of recessive deleterious mutations from mating among genetically related individuals in small populations (Charlesworth and Willis 2009).

Populations less than approximately 100 individuals also face extinction risk from **demographic stochasticity**. This arises from fluctuating sex ratios, random variation in the number of offspring produced, and chance mortality events (Lee et al. 2011). In large populations, such randomness is averaged out across individuals and has no discernible effect. But when only a small number of individuals remain, chance occurrences, like a string of all male offspring, can significantly affect demographics.

Finally, in some species, extinction dynamics are influenced by **Allee effects**, which are cooperative group processes that facilitate population growth (Boukal and Berec 2002). Examples include communal defence against predators, communal raising of offspring, and efficient mate finding. When populations that depend on Allee effects become too small for group processes to operate, population growth rates can quickly decline, reducing viability.

Box 6.4. Extinction Vortices: the Case of the Heath Hen

The various demographic and genetic processes that affect small populations often operate synergistically, causing an **extinction vortex**. The extinction of the heath hen, as recounted by Shaffer (1981), provides an illustrative example. The heath hen was a type of prairie chicken originally found in the northeastern US. Once common, its abundance steadily declined with European settlement as a result of habitat loss and increased mortality (i.e., systematic decline). By 1876, the species remained only on Martha's Vineyard, and by 1900 there were fewer than 100 survivors. In 1907, a portion of the island was set aside as a refuge for the birds, and a program of predator control was instituted. The population responded to these measures and by 1916 had reached a size of more than 800 birds. But in that year, a fire (natural catastrophe) destroyed most of the remaining nests and habitat, and during the following winter, the birds suffered unusually heavy predation from a high concentration of goshawks (environmental stochasticity). The combined effects of these events reduced the population to 100–150 individuals. In 1920, after the population had increased to about 200, disease (environmental stochasticity) took its toll, and the population was again reduced below 100. In the final stages of the population's decline, the birds appeared to become increasingly sterile, and the proportion of males increased (demographic stochasticity and inbreeding). By 1932, the species was extinct.

Spatially Structured Populations

To this point, we have implicitly assumed that populations exist in simple homogenous landscapes. We will now consider the dynamics of populations in more realistic environments, where habitat quality varies across space.

In a heterogeneous landscape, the various limiting factors that influence reproduction and mortality will vary from location to location. Thus, population densities will also be spatially variable. In the absence of immigration, we can expect populations to persist only in areas where the carrying capacity is greater than zero.

Population dynamics are more complex when immigration and emigration are included in the system. We must now differentiate between **source habitats**—areas where intrinsic growth is sufficient to maintain viability—and **sink habitats**—areas where populations are only able to persist through immigration from source populations (Fig. 6.9; Pulliam and Danielson 1991). Though sink habitats cannot support populations independently, they still contribute to overall species viability. They expand the range of the species and increase overall abundance, both of which help to buffer against environmental stochasticity (Carroll et al. 2003).

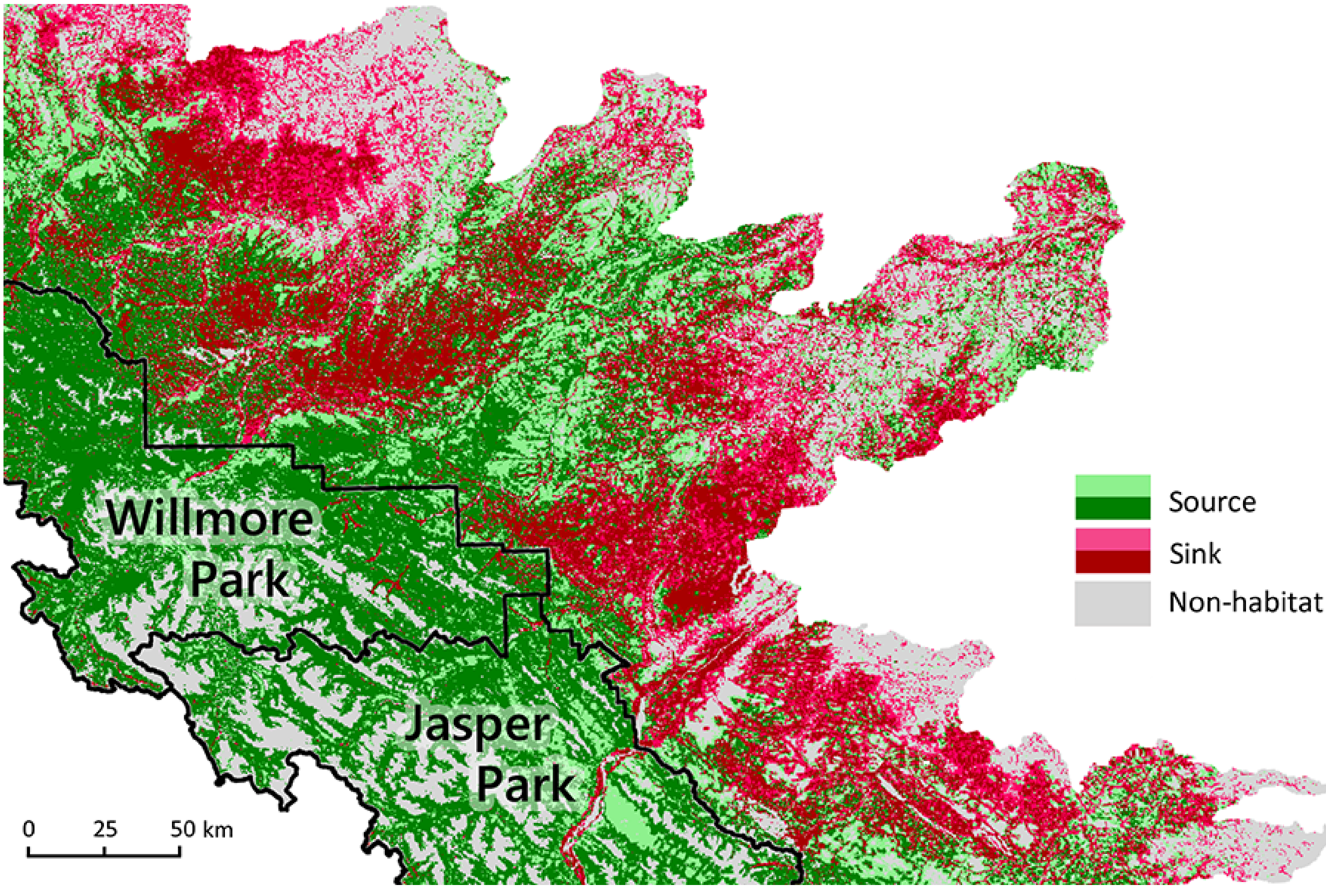


Fig. 6.9. The distribution of source and sink habitat for grizzly bears along the eastern slopes of the Rocky Mountains in Alberta near Jasper National Park. In this case, sink habitats east of the mountain parks have an unsustainably high rate of mortality associated with high levels of human access. Adapted from Nielsen et al. 2006.

For some species, habitat is perceived as discrete patches. For example, the scattered sloughs in a prairie landscape constitute discrete habitat patches for frogs and other wetland species. If the patches are sufficiently isolated from each other, a population may be divided into subpopulations that exhibit semi-independent dynamics. In this case, the assemblage of subpopulations is referred to as a **metapopulation** (Hanski 1998).

If the patches of habitat are relatively small, the individual subpopulations may be vulnerable to decline or extinction from the stochastic and genetic processes we discussed earlier (Hanski 1998). But there also exists the possibility for rescue through emigration from neighbouring subpopulations. The overall metapopulation is said to be in an equilibrium state when the processes of loss and rescue are in balance. The main parameters influencing these dynamics include:

- Patch size. Large patches are more stable and produce more migrants.
- Degree of environmental synchrony among patches. The potential for rescue improves when patches do not experience the same pattern of fluctuations.
- Dispersal ability. Movement is affected by species-specific dispersal traits and by the permeability of the non-habitat matrix.
- Distance between patches. Longer distances reduce the number of immigrants, but also reduce the level of

environmental correlation, so the effects on viability are complex.

In natural landscapes, species that exhibit metapopulation dynamics normally have sufficient dispersal ability to keep subpopulation decline and rescue in balance and to maintain gene flow. Again, natural selection has long ago weeded out species unable to do so. However, anthropogenic disturbance can easily disrupt this equilibrium, either by reducing the viability of individual subpopulations or by changing the permeability of the matrix (Mennechez et al. 2003).

Human development also can produce metapopulations artificially, by fragmenting the habitat of previously continuous populations (Fig. 6.10). This has occurred extensively in the Agricultural South (Fig. 5.5). The viability of such nouveau metapopulations is not guaranteed because connectivity among remnant patches may be low under such artificial conditions, at least for some species (Tucker et al. 2018).

Fig. 6.10. Metapopulation dynamics can arise when habitat is fragmented through human land use. In this agricultural landscape in southern Ontario, only scattered patches of the original forest remain. Credit: Google.

Species Range

The range of a species is the overall geographical area in which its populations are found. In the simplest case, where limiting factors exist as uniform gradients, we would expect a species to be most abundant in the centre of its range and progressively decline toward the periphery. However, limiting factors in real landscapes often exhibit non-uniform patterns; therefore, species distributions are typically quite complex (Sexton et al. 2009; Boakes et al. 2018). Gaps in distribution may exist within the range, and the centre need not have the highest abundance (Gaston 2009).

Range boundaries are also influenced by the interplay between dispersal and environmental stochasticity (Hargreaves et al. 2014). If environmental variability is low, species may routinely occupy sink habitat in peripheral regions through ongoing emigration from source habitat. Conversely, in the face of high environmental stochasticity and low dispersal ability, areas where the carrying capacity is only marginally positive may be unoccupied. Because of these processes, fragmentation of population structure is common along range margins.

Limiting factors may systematically change over time, redefining the region in which populations can persist. Human disturbances have been the main driver of such systematic changes, leading to widespread range contractions in many species (Channell and Lomolino 2000). More recently, ranges have begun to shift as a result of climate change. In this case, all species are affected (see Chapter 9).

When environmental conditions undergo systematic change (regardless of the cause), population responses may lag behind. Disequilibrium is most likely to occur in species with long generation times (e.g., trees) and when the pace of environmental change is rapid. In the case of deteriorating conditions, demographic lags may result in populations that continue to exist while on an extinction trajectory. Such populations represent an **extinction debt** (Kuussaari et al. 2009).

Tactical Modelling

The general principles that have emerged through strategic modelling form the foundation of species-level conservation. But broad concepts and principles can only go so far. Applied conservation also requires attention to the unique characteristics of individual populations in real landscapes, especially for the assessment of risk and the efficient allocation of conservation efforts. This is the purview of field research and associated tactical models. Not all conservation practitioners will be directly involved in conducting field studies and modelling, but most will be users of the information provided and need to be able to assess it critically and apply it effectively.

Most tactical models are statistical models that mathematically summarize important patterns and relationships in the data generated by individual observational studies. Tactical models can also be constructed by combining the findings from multiple observational studies into a composite process-based model. Both approaches are used in applied settings to generate predictions that support management decisions. These models also can identify the factors that are most influential in a system and pinpoint where key uncertainties lie. This information provides guidance for future field research and helps managers understand the reliability of the predictions being made.

There are three main categories of tactical models (Fig. 6.11):

1. **Habitat models**, for the study of habitat selection and species distribution
2. **Population models**, for the investigation of population dynamics and viability
3. **Landscape models**, for the study of connectivity patterns and the exploration of population responses to landscape changes from anthropogenic disturbance and climate change

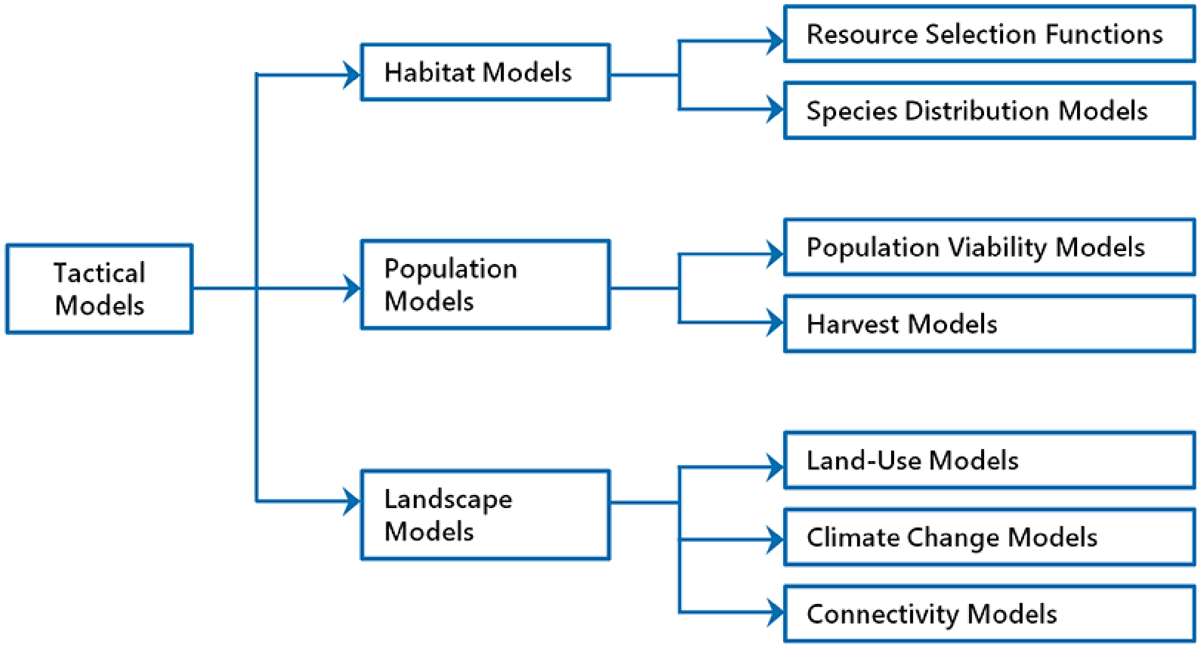


Fig. 6.11 A taxonomy of the most common types of tactical models used to support conservation at the species level. In practice, hybrid models are common.

In practice, hybridization among these model categories is common, and arguably essential in many applications. But for the sake of clarity, we will treat the various modelling categories as discrete entities, emphasizing the intended purpose of each approach and its main assumptions and limitations. In the case studies presented in Chapter 11, we will see how tactical models are used in real-world applications.

Habitat Models

Habitat models are used to gain insight into the basic ecology of a species and to make predictions that inform management. At the local scale, the main interest is identifying the habitat types that are most important to a species so they can be maintained through management prescriptions and guidelines.

At broader scales, the main interest is in identifying geographic regions that feature high-quality habitat. This information is used to select optimal sites for species reintroductions and for the establishment of protected areas (e.g., Nielsen 2006). It also can inform the designation of critical habitat under SARA (see below). Habitat studies are also integral to determining range extent and range trends, which are often incorporated into status assessments and management targets (Loehle and Sleep 2015).

Most habitat modelling involves the statistical extrapolation of data collected in observational field studies (Lele et al. 2013). Statistical approaches are used because it is generally not feasible to study all individuals in a population. We sample a subset of the population and then use models to infer habitat relationships for the entire population.

The data for statistical habitat models are collected in a variety of ways. A common approach is to survey usage along transects or at specified locations. Usage is determined by visual sighting (e.g., aerial surveys of large mammals), auditory detection (e.g., breeding bird surveys) and evidence of recent use (e.g., snow tracking). In recent years, camera traps, acoustic recorders, and DNA markers have greatly augmented the collection of survey data (Steenweg et al. 2017).

Fig. 6.12. A wolf fitted with a radio collar as part of a telemetry study in Yellowstone National Park. Credit: W. Campbell.

Another common approach for collecting usage data is to track the movement of individuals using attached location transmitters (Fig. 6.12). In the past, these transmitters emitted radio signals that required manual relocations to be made using portable antennas. More recently, tracking devices have been developed that automatically take GPS locations at timed intervals, providing much greater detail on movement paths (Hebblewhite and Haydon 2010).

To determine habitat associations, data also must be collected on relevant environmental variables. Our ability to do this effectively has undergone tremendous improvement over the last two decades, with the advent of geographic information systems (GIS) and multi-layered spatial datasets. By documenting the distribution of habitat types across the entire study area, it is possible to determine not only which habitat types are being used, but which are being actively selected. We say that a habitat type is being selected if its proportional use exceeds its availability in the study area (Aarts et al. 2012; Boyce et al. 2016).

Analytical approaches have evolved alongside advances in data collection and data processing. The relevant approaches fall into two main categories. **Resource selection functions** (Lele et al. 2013) explain and predict habitat selection by individuals at the local scale, and **species distribution models** (Elith and Leathwick 2009) explain and predict patterns of species occurrences at broad scales. The statistical methods used by these two approaches are broadly shared, so it is mainly scale and purpose that differentiate them (Aarts et al. 2012). Newer approaches, such as step selection functions, have been developed to handle the large amounts of data being generated in GPS-based tracking studies (Thurfjell et al. 2014).

An older form of habitat modeling, widely used in the twentieth century, is **habitat suitability index modelling**, a quasi-mechanistic approach (Schamberger et al. 1982). The development of these models begins with a review of the literature to identify the habitat variables most strongly related to population dynamics of the species of interest. Using expert opinion, these variables are then qualitatively combined into an equation that provides an index of habitat quality. Today, quantitative statistical approaches are preferred; however, habitat suitability index modelling still has a role in cases where the detailed datasets needed for quantitative analysis have not or cannot be collected (e.g., Stevens et al. 2008).

All forms of habitat modelling are subject to various limitations that users of the results need to consider. There are three main areas of concern: reliability of estimation, differences between habitat selection and habitat importance, and inappropriate extrapolation.

Problems with reliability arise from the challenges inherent in sampling biological systems. Depending on the species, habitat associations may vary by age, sex, season, and year (Alldredge and Griswold 2006). Relationships also can be affected by habitat availability and its distribution, since animals can only select from what is on offer (and this might not include what is most desired). In addition, observed associations can be influenced by detection biases and by errors in the classification of environmental variables (Wilson et al. 2005; Burton et al. 2015). The latter is a common problem when satellite imagery is used to specify vegetation types (Rocchini et al. 2013). If these potential sources of variability and bias are not addressed through appropriate sampling design (MacKenzie and Royle 2005), the model will not provide a reliable representation of the system.

A second limitation of habitat models is that selection is not necessarily indicative of importance to demography (Alldredge and Griswold 2006). Much depends on how the environmental variables are specified. If habitat variables are poorly defined, important relationships may be overlooked. Furthermore, some critical habitats, such those required for nesting, may only be used for limited periods, which can lead to an underestimation of their importance (Alldredge and Griswold 2006).

At broader scales, the interpretation of selection studies may be complicated by demographic processes (Boyce et al. 2016). High-quality habitats might be unused for reasons unrelated to habitat conditions, such as overhunting. Conversely, immigration into sink habitats may result in an overestimation of their value. Consequently, the interpretation of habitat value from selection studies should be done in conjunction with knowledge about the basic ecology of a species.

Finally, as with all statistical models, there are limits to extrapolation. Predictions of habitat use applied within the study area itself are usually sufficiently reliable for decision making as long as no major changes occur in the system (Elith and Leathwick 2009). However, when extrapolating to external regions or to future periods, reliability is unknown.

A key concern is that habitat availability often varies among regions and time periods, and this can influence selection patterns. For example, Canada warblers in New Hampshire select red maple swamp, dominated by red maple, balsam fir, and red spruce, whereas those in the western Canadian boreal forest select old-growth aspen and aspen-spruce mixedwoods (EC 2016). Choices are also influenced by biotic interactions, which may not be constant over space and time (Elith and Leathwick 2009). In the future, as temperatures rise, different rates of northward migration may result in novel species interactions that influence competition and habitat use in new ways (Williams and Jackson 2007).

One solution for predicting habitat selection across larger areas is to expand the study area and another is to create separate regional models (Wiens et al. 2008). The latter is more common because it does not require any sacrifice of detail, and because smaller studies are easier to fund and coordinate. As for modelling future periods, we will discuss the relevant approaches in Chapter 9, in the context of climate adaptation.

Box 6.5. Ecological Niche

A species' niche can be defined as the range of biotic and abiotic conditions within which its populations can persist without immigration (Araujo and Guisan 2006; Wiens et al. 2010). The conditions being referred to here are related to the limiting factors for population growth we discussed earlier, indicating a direct connection between ecological niche and demography.

There is also a connection between niche and habitat. Niche refers to the set of conditions that support existence, and habitat refers to the physical locations where those conditions can be found. This distinction is fairly nuanced, and consequently the terms are often used interchangeably (Whittaker et al. 1973).

A species' tolerance to individual limiting factors, such as temperature, will often be broader than its observed distribution suggests. This is because a species will only occur where all of its essential parameters are within acceptable limits. Moreover, distributions are strongly influenced by competition and other biotic interactions. Therefore, it is useful to differentiate between the fundamental niche of a species, referring to its complete range of tolerances, and the realized niche, which is what we actually observe. Here again, there is a lack of consistency in the criteria used for making this distinction (Araujo and Guisan 2006).

In conservation applications, we are mainly concerned with the realized niche. Strictly speaking, the realized niche includes anthropogenic disturbances as a limiting factor. However, in practice, it is common to differentiate the ecological niche that exists under natural conditions from the niche that is realized in landscapes impacted by humans.

The term "ecological niche" is also used to describe the functional role of a species within a community, particularly in relation to its food and enemies. In this usage, a species' niche is seen as a collection of structural, physiological, and behavioural adaptations that has evolved in the context of competition with other species. This form of niche is associated with the concept of competitive exclusion, which posits that no two species can indefinitely occupy the same ecological niche, driving evolutionary differentiation (Levin 1970).

Population Models

Tactical population models were initially developed to support the management of species that are harvested. The core concept in this application is **maximum sustained yield** (Sissenwine 1978), in which harvest rates are tailored to achieve a population density that delivers maximal population growth (i.e., the top of the curve shown in Fig. 6.2). Such models have been used to predict the maximum sustained yield under varying environmental conditions and alternative harvest scenarios (Taylor et al. 2008).

Tactical population models are also used in the management of species at risk (Boyce 1992). Here, the emphasis is on modelling population trends with the aim of characterizing the risk of extinction, prioritizing threats, exploring alternative management approaches, and identifying key uncertainties.

In population models, abundance is a function of the population birth rate and the population death rate, often broken down by age class and sex. These two core variables are in turn influenced by other factors, such as habitat protection measures and artificial rearing. The choice of model components is based on what is relevant to the species of interest, constrained by the availability of information. In most cases, a link to habitat is warranted, but it is not always the most important factor. Knowledge gaps often stimulate additional field research, leading to later revisions of the model and evolving model complexity.

Model construction begins with a conceptual model that maps out the main components of the system and qualitatively describes the linkages between them. This conceptual model is then typically implemented as a computer simulation, which permits rapid exploration of the system's behaviour. To run the model, the variables are parameterized using data from relevant field studies. Many models allow parameter values to fluctuate at each time step (within natural bounds) to simulate stochastic processes. Because each run produces a different result, the behaviour of stochastic models must be summarized over hundreds or thousands of runs.

As output, population models provide the predicted size of the population over time. In many cases, the raw population trend is itself of primary interest. In other cases, the emphasis is on population viability, expressed as the mean time to extinction across all runs. Viability is sometimes also expressed as the **minimum viable population size**, which is the minimum number of individuals required to achieve a specified probability of persistence (Shaffer 1981). This amounts to an expression of acceptable risk and requires a time period to be specified. For example, we might seek a population that has a 99% chance of surviving for the next 100 years.

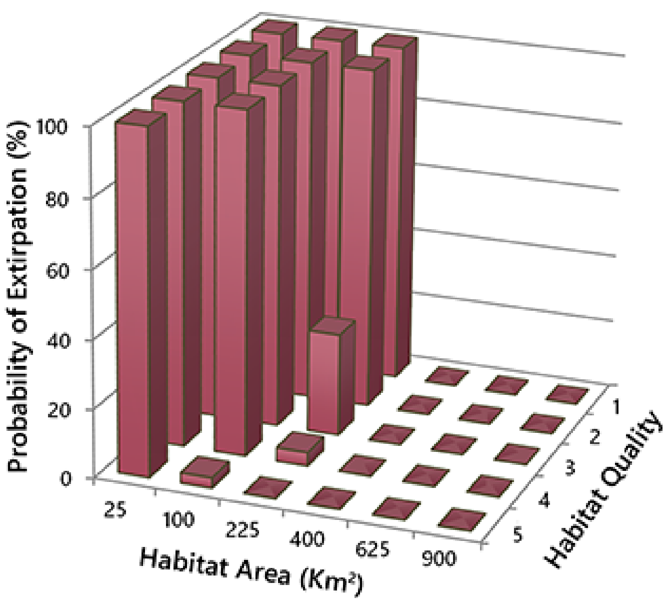


Fig. 6.13. Population viability analysis results for the threatened Newfoundland marten population. The results illustrate the effect of habitat area and quality on the probability of marten extirpation. Adapted from Schneider and Yodzis 1994.

In practice, population modelling is often exploratory. Parameters under management control may be run at different levels, corresponding to alternative management strategies. For example, the relationship between population trends and habitat supply might be explored by varying the amount of habitat available to the species (Fig. 6.13). Because parameter values are rarely known with precision, it is common to run the model across a range of likely values. This is known as **sensitivity testing**. Such testing is useful for quantifying uncertainty and identifying the parameters that are most influential. Parameters that are both influential and poorly characterized are a priority for future research.

As with habitat models, population models are subject to limitations that end-users should be aware of (Flather et al. 2011). To begin, the relationships in population models are based on correlative field studies and are subject to all the limitations of these kinds of studies.

More generally, even the most complex population models suffer from missing relationships and other knowledge gaps. Ecological data are difficult and costly to obtain, so only a subset of the factors and relationships that affect a population can be incorporated. Modellers must work with what they have and hope that the missing relationships have limited effect or are accounted for indirectly. However, it is impossible to know this with any certainty.

Given these limitations, we should not expect population models to be omniscient predictors of the future (Boyce 1992). The quantitative estimates of population size they provide are often just coarse approximations, especially in species at risk, which are generally subject to major knowledge gaps (Fieberg and Ellner 2000; Morrison et al. 2016). The main utility of these models is in providing insight into the relative importance of the threats facing a species, helping managers allocate limited conservation resources efficiently (Boyce 1992). In addition, meaningful comparisons can often be made among alternative management approaches, aiding in the identification of the most effective approach. In summary, population models allow us to consolidate what we know about a system and thereby inform management decisions, but we must remain attuned to their limitations.

Landscape Models

Landscape models are used to explore landscape connectivity and landscape dynamics. These models incorporate elements of both habitat and population models and can be thought of as an extension of these other approaches.

Landscape **connectivity models** are used to predict how individuals move within a given landscape. Management applications are typically focused on identifying critical movement corridors so they can be protected (see Chapter 7).

Connectivity modelling usually begins with the production of a so-called "resistance" layer, which characterizes the permeability of a given landscape to animal movement (Rudnick et al. 2012). The resistance layer is generated within a GIS by assigning different levels of permeability to landscape patches or pixels based on their composition and structure. Permeability rules may be derived from models of habitat suitability, gene flow, or individual-based movement (Rudnick et al. 2012). Alternatively, if the data required for detailed modelling are not available, permeability may be based on the level of habitat disturbance (Koen et al. 2014a).

With a resistance layer in hand, various techniques are available to identify optimal movement pathways, as well as movement pinch points that are priorities for management intervention (Rudnick et al. 2012). Least-cost path modelling has been commonly used for this purpose in the past (Rouget et al. 2006; Poor et al. 2012). Newer methods are also being employed, such as circuit theory, in which animal movements are simulated as an electric current in a resistance network (Pelletier et al. 2014; Proctor et al. 2015).

The development of a resistance layer invariably involves simplifications because detailed information on movement behaviour is difficult to obtain and because there are practical limitations on model complexity. In most cases, the resistance layer provides only a coarse approximation of how organisms move through real landscapes (Rudnick et al. 2012). For many applications, such as the identification of critical movement bottlenecks, this may suffice.

Landscape models are also used to explore the effects of landscape change on population dynamics. Some applications focus on individual species and can be considered enhanced versions of the population models we discussed earlier. The purpose of adding landscape dynamics is to increase model realism. In other applications, the emphasis is on modelling landscape changes, with the aim of supporting land-use decision making. These types of models usually incorporate sub-models to track the effects of landscape change on selected output indicators. The sub-models used to predict focal species responses are typically fairly simple.

Landscape models are also used to estimate the **minimum viable area** of a species, representing the geographic correlate of minimum viable population size (Brito and de Viveiros Grelle 2004). These models are subject to the limitations of population viability modelling discussed earlier.

Finally, landscape models are used to predict the effects of climate change on populations. Most applications involve bioclimatic envelope models—a form of species distribution model—to predict range shifts under alternative climate warming scenarios. Less commonly, mechanistic models are used to investigate transitional processes. We will discuss these types of models in Chapter 9.

Recovery Planning

Planning is a central element of conservation. This is where social values and scientific knowledge come together to inform decisions about what we will do and where we will do it. It is here that the broad goal of maintaining biodiversity is translated into operational terms.

In this chapter, we will focus on planning as it pertains to the management of species at risk in Canada. The emphasis will be on the mechanics of the process and the practical challenges inherent in moving from policy to practice. We will touch on planning again in Chapter 7, in the context of integrated landscape management, and in Chapter 10, in the context of structured decision making. Chapter 11 provides some practical examples of species recovery planning.

As recounted in Chapter 3, the federal and provincial governments share responsibility for managing species at risk. Although SARA is national in scope, it only applies directly to areas of federal jurisdiction, including the territories, oceans, national parks, some freshwaters, and most migratory birds. The primary responsibility for managing terrestrial species falls to the provincial governments, under provincial species at risk legislation and other wildlife laws. In practice, the federal and provincial governments coordinate their efforts, albeit somewhat uneasily. Provincial cooperation is motivated by the fact that SARA includes “safety net” provisions that come into effect if a province fails to meet the protection standard prescribed by SARA (GOC 2002, Sec. 80.4). The following discussion of species at risk management is based on the federal process.

Overview of SARA

Under SARA, the key steps in the recovery process, as well as timelines, are explicitly defined in law, distinguishing SARA from most other Canadian conservation legislation. The process begins with an assessment by the independent Committee on the Status of Endangered Wildlife in Canada (COSEWIC). COSEWIC commissions status reports on potentially at-risk species and then provides a status recommendation to the federal government, patterned on an international framework (Mace et al. 2008). The categories are: extinct, extirpated, endangered, threatened, of special concern, and not at risk. The government has nine months to either accept a recommendation (i.e., to “list” it), decline it with cause, or return the issue to COSEWIC for further clarification (GOC 2002, Sec. 27.1).

The intent of this two-step listing process is to separate the scientific and political aspects of conservation decision making. COSEWIC’s assessments are to be based strictly on the biology of the species, leaving the consideration of feasibility, cost, and social and political ramifications to elected officials (ECCC 2016a).

Species that are subject to commercial harvest and Indigenous harvest (particularly in Nunavut) have often been denied listing on socio-economic grounds (Findlay et al. 2009). Marine fish have been the most affected. A review of COSEWIC listing recommendations for these species found that 71% were denied (McDevitt-Irwin et al. 2015). According to Schultz et al. (2013), the bias against marine species is not just related to economic impacts but also reflects the fact that ocean wildlife is under federal jurisdiction (i.e., the implications of listing hit closer to home). Such systematic circumvention of SARA is contrary to what was intended by lawmakers when they passed the Act.

Once a species is listed in the SARA registry, a **recovery strategy** must be prepared within one year if the species is endangered, and within two years if the species is threatened (GOC 2002, Sec. 42.1). For species of special concern, a management plan is required within three years, and the emphasis is on preventing further declines rather than on recovery. The planning process is overseen by the relevant federal agency: Fisheries and Oceans Canada for aquatic species, Environment and Climate Change Canada for terrestrial species, and Parks Canada for species within national parks. Strategies are developed by species-specific recovery teams composed of government staff and external experts.

Recovery strategies are largely science based, drawing on published reports and the knowledge of domain experts. Strategies follow a standard format that includes the following elements:

- Species biology, including distribution, population size and trends, habitat needs, and limiting factors
- Threat analysis, with at least a qualitative ranking of importance
- Assessment of whether recovery is feasible, accounting for reproductive capacity, habitat availability, potential for threat mitigation, and existence of practical recovery techniques
- Recovery goal and objectives
- Recovery approaches
- Identification of critical habitat
- Knowledge gaps and priorities for future research

After a recovery strategy is completed, the federal government must develop an **action plan**, in consultation with stakeholders. Action plans define the management steps that will be taken to achieve the objectives outlined in the recovery strategy (GOC 2002, Sec. 49). It is at this stage that socio-economic factors are considered, rather than in the recovery strategy. The government must monitor the action plan and report on progress toward meeting recovery objectives five years after the plan comes into effect (GOC 2002, Sec. 55).

Recovery planning is subject to several limitations (SPI 2018). Species at risk are notoriously difficult to study, and funding for research is limited. Species like woodland caribou and killer whales, which have been studied intensively, are the exceptions rather than the rule. For most species at risk, recovery teams have only basic information on distribution and threats to work with, and little or no tactical modelling.

There are also process-related problems to contend with. As previously noted, many species are denied listing and never even enter the planning process (McDevitt-Irwin et al. 2015). For listed species, planning is frequently subject to delays at both the recovery strategy stage and the action planning stage (AGC 2013). Moreover, most recovery strategies to date have not identified critical habitat as required (Bird and Hodges 2017). Matters have improved in recent years, following several successful court challenges (FCC 2014). However, none of the process deficiencies have been fully remedied.

Finally, the mechanisms for plan implementation are poorly developed and many plans languish at this stage. SARA acknowledges the need to consider other social objectives but does not describe how trade-off decisions are to be made. Implementation is also hindered by the split between federal and provincial responsibilities (i.e., federal planning and provincial implementation).

No unifying framework exists for managing focal species that are not listed as species at risk. For harvested species, policies and plans are usually developed by fish and wildlife departments and forestry departments at the regional or provincial scale. These plans reflect local interests and priorities. In some cases, the aim is to maintain historical population levels, as described by NRV. More often, the objective involves some combination of ecological and social outcomes, taking the costs and benefits of management actions into account. Species that are not harvested generally do not have formal management plans and are instead managed through landscape-scale conservation approaches.

Incorporating Genetic Diversity

Different geographic regions exert different selective pressures. Consequently, wide-ranging species often exhibit local genetic adaptations. These adaptations are not always obvious but can be demonstrated through transplant experiments. For example, aspen from Minnesota exhibit almost twice the biomass growth of local aspen when transplanted to Alberta (Schreiber et al. 2013). In turn, Alberta aspen exhibit traits, such as late bud break, that are protective against exceptional spring frosts that occur sporadically in Alberta but not in Minnesota (Li et al. 2010).

The development of local adaptations is counteracted, to a variable degree, by dispersal, which tends to homogenize populations (Fraser et al. 2011). The development of local adaptations also depends on how much environmental variability a species experiences. Consequently, the degree of spatial structuring varies among species. Some species feature genetically distinct populations with low rates of genetic mixing, whereas other species exhibit genetic homogeneity over large geographic regions (Laikre et al. 2005). Intermediate forms, like aspen, may exhibit continuous genetic change over geographic distance (Fig. 6.14).

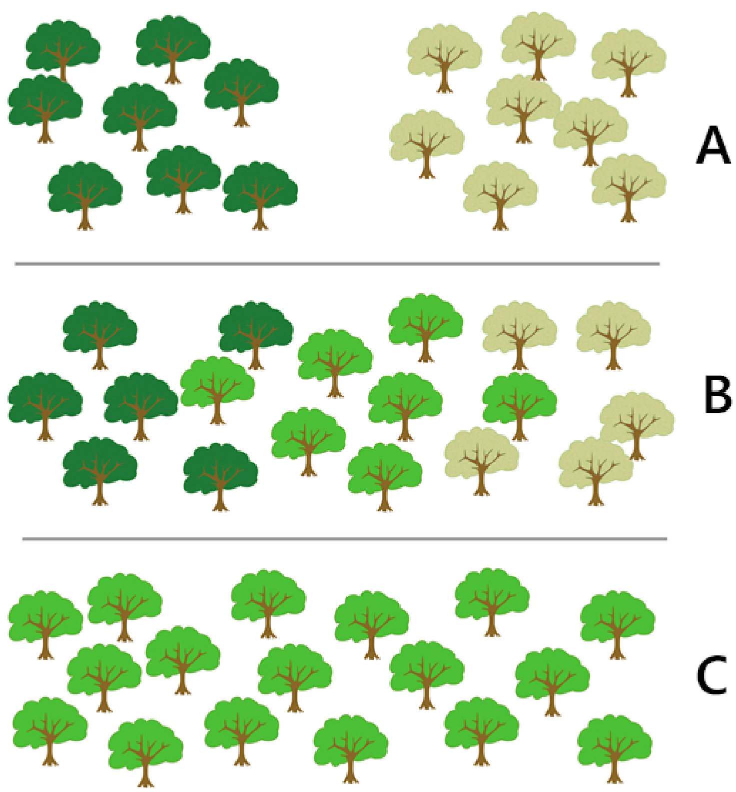


Fig. 6.14. The genetic structure of populations can be crudely classified into three basic types. A) Distinct populations, where gene flow is low enough for genetic divergence. B) Continuous change, where genetic composition changes continuously over space, resulting in isolation by distance. C) No differentiation, where gene flow is so extensive that genetic homogeneity prevails across the entire region. Adapted from Laikre et al. 2005. Clipart by X. Dengra.

Local genetic adaptations are important to conservation because they maximize the **fitness** of individual populations and they provide genetic variability for adapting to change at the species level (Lande and Shannon 1996).

The challenge conservationists face is determining the appropriate genetic unit for protection (Waples 1998). The basic objective is to identify subsets of a species that have genetic attributes important to overall species viability. These are referred to as "evolutionary significant units" (Ryder 1986). The difficulty is in establishing the link between genetics and species viability and in determining the appropriate level of importance (Waples 1998).

In Canada, COSEWIC uses a set of guidelines (Green 2005) to identify so-called "**designatable units**" below the species level, on a case-by-case basis. The main criteria include disjunct range, occupancy of different ecoregions, and genetic distinctiveness. In each case, differentiation is made only if the individual units also differ in conservation status. If the units share the same status, there is no practical value in distinguishing them because they will receive the same management treatment.

As with any such system, COSEWIC's approach to selecting designatable units has shortcomings. Genetic sampling across broad regions is difficult and expensive, even with indirect sampling techniques (e.g., using hair and scat samples). As a result, only a relatively small number of species have been well studied. Moreover, the relationship between genetic variability and fitness is poorly understood, and quantitative analysis is generally not possible (Green 2005). In practice, COSEWIC bases its decisions on expert opinion, and most designatable units are defined very coarsely (e.g., broad regional populations such as "eastern" and "boreal").

There are no formal mechanisms or requirements under SARA for maintaining genetic diversity below the level of designated units. Therefore, the consideration of fine-scale genetic structure in recovery plans is quite variable. For species reduced to very small populations, the maintenance of all remaining genetic variance and the avoidance of inbreeding are often identified as major management concerns, particularly when artificial breeding programs are used. For other species, genetics may be addressed indirectly, through recovery objectives that seek to maintain the species across its entire range. Some recovery strategies make no reference to genetics at all.

For focal species not listed under SARA, the consideration of genetics is generally quite limited, with certain exceptions. For example, replanting programs (e.g., commercial trees) and restocking programs (e.g., sport fish) often include measures to ensure that local genotypes are maintained. In addition, management plans for species with highly fragmented populations may include measures to avoid inbreeding. For most other species, genetic considerations are handled indirectly through distribution objectives, if at all.

Setting Objectives

The articulation of objectives is central to the planning process—it is what motivates and directs management action. For species at risk applications, objective setting must account for differences in the status of individual species. What needs to be done, and what is feasible to do, is much different for a species on the precipice of extinction as compared to a widely distributed species that is undergoing a slow decline. SARA defines a hierarchical objective composed of three elements: (1) to prevent wildlife species from becoming extinct, (2) to provide for the recovery of wildlife species that are threatened as a result of human activity, and (3) to manage species of special concern to prevent them from becoming threatened (GOC 2002, Sec. 6).

Under SARA, recovery is a multistage process (Fig. 6.15). For species on the edge of extinction, the emphasis is on ensuring survival in the immediate future. Addressing the compound risks associated with small population size is paramount. Once immediate survival is assured, the focus shifts to securing long-term recovery. The aim of recovery is to "return the species to whatever its natural condition was in Canada prior to being put at risk by human activities" (ECCC 2020). This entails additional population growth, range expansion, and the establishment of multiple distinct populations. The long-term recovery goal serves as a beacon, providing context and direction for the overall recovery process.

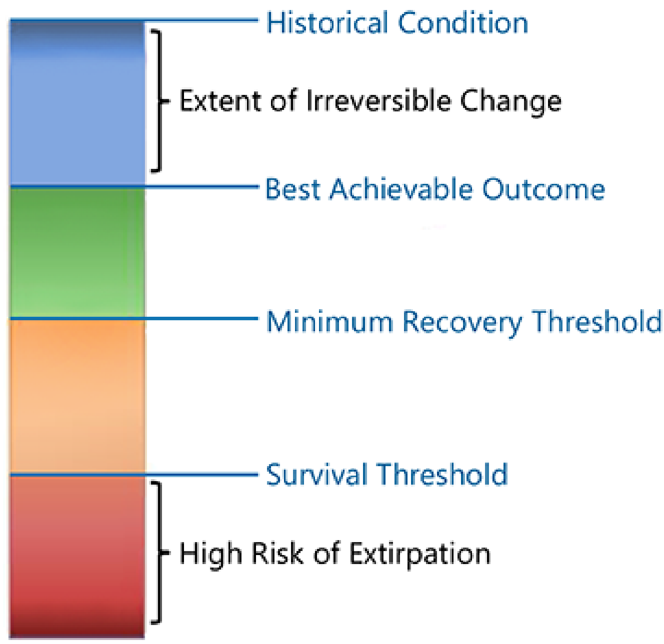


Fig. 6.15. Under SARA, species recovery is seen as a multistage process (ECCC 2020). The long-term aim is to restore the species as close as possible to its historical state.

For many species, the historical condition is not achievable because of irreversible ecological changes. In such cases, the recovery endpoint is the best outcome that is biologically and technically achievable (Fig. 6.15). Socio-economic constraints also affect recovery outcomes, but they are addressed later in the planning process, not when setting biotic objectives (ECCC 2020).

Policymakers have not yet determined how to accommodate climate change when setting species recovery objectives. Using historical conditions as the reference state remains serviceable for the time being because climate-induced ecological changes are not yet widespread. But going forward, we will need to transition to a dynamic reference state that is robust to climate change. We will explore the options for doing so in Chapter 9.

Identifying Critical Habitat

SARA stipulates that recovery strategies must define critical habitat, but this has proven to be challenging (Bird and Hodges 2017). The process has suffered from protracted delays, non-compliance, and a lack of policy guidance (Martin et al. 2017). New policy has recently been developed to provide additional direction (ECCC 2016b), and compliance is now improving (Bird and Hodges 2017).

The process of defining critical habitat under SARA has three steps: (1) assemble and assess the available data; (2) identify critical habitat, including its geographic location and biophysical attributes; and (3) describe the specific threats to the identified habitat. It is acknowledged that data inadequacies will exist for most species at risk. SARA policy (ECCC 2016b) suggests that recovery teams should generally do the best they can with existing data:

> The fact that there will be better and/or more information on which to base critical habitat identification at some point in the future cannot be used as a reason to delay identifying critical habitat to the extent possible, based on the best information available at this time. If all of the critical habitat cannot be identi-

fied based on the best available information, then critical habitat will be identified to the extent possible. … Critical habitat identification is often an iterative process and partial identification may be possible in advance of full identification. (pp. 12–16)

When identifying critical habitat, recovery teams are guided by SARA's definition: "critical habitat means the habitat that is necessary for the survival or recovery of a listed wildlife species" (GOC 2002, Sec. 2). As previously noted, socio-economic implications are intended to be addressed at the action planning stage, rather than when critical habitat is being identified (ECCC 2016b).

In most recovery strategies to date, critical habitat has been identified on the basis of known use (Camaclang et al. 2015; Martin et al. 2017). For example, critical habitat for the loggerhead shrike (*migrans* subspecies) is identified as patches of suitable grassland occupied by a confirmed breeding pair during the period 1999–2008, along with any other suitable patches within 400 m of a nesting site (EC 2015). A shortcoming of this approach is that it provides no assurance that there will be sufficient habitat for species survival, let alone recovery (Camaclang et al. 2015; Martin et al. 2017). The survival of many species at risk requires populations to increase from critically low levels, with a concomitant need for habitat that is currently unoccupied. Habitat requirements for full recovery are even greater.

The upshot is that there still is a gap between policy and current practice. Critical habitat is being defined very narrowly (if at all) and generally supports only immediate needs rather than long-term survival and recovery. This may improve over time, as additional field studies and modelling enable the identification of critical habitat on the basis of population recovery needs rather than current use. But progress will be slow unless additional staff and research funding are brought to bear (Bird and Hodges 2017; Martin et al. 2017).

In the final step, recovery teams must characterize the threats to critical habitat, providing the basis for protection measures. Threats include all activities that may degrade critical habitat, either permanently or temporarily, "such that it would not serve its function when needed by the species" (ECCC 2016b, p. 22). Detailed assessments are needed because, in practice, protection measures most often involve the mitigation of specific threats rather than the establishment of formal protected areas.

For species of special concern and other focal species, there is no legal requirement for identifying critical habitat. Therefore, other mechanisms must be used to identify and protect the habitat of these species. This usually entails integrating focal species management objectives into broader land-use planning programs and protected area initiatives, which we will discuss in subsequent chapters.

Taking Action

Under SARA, the implementation aspects of recovery planning are captured in action plans, which must include the following components (GOC 2002, Sec. 49.1):

- A statement of measures to address threats and achieve recovery objectives, as well as an indication of when those measures will take place
- A statement of measures to protect critical habitat, to the extent that it has been identified
- Methodology for monitoring and reporting on the recovery of the species
- An evaluation of the socio-economic costs of the action plan and the benefits to be derived from its implementation

Although SARA places explicit prohibitions on the destruction of critical habitat as well as on the killing, harming, and capturing of listed species (GOC 2002, Sec. 32 and Sec. 58), these are not ironclad assurances of protection. SARA allows companies and individuals to apply for permits that allow some incidental harm, so long as it does not "jeopardize the survival or recovery of the species" (GOC 2002, Sec. 73).

Furthermore, SARA's prohibitions only directly apply in areas of federal jurisdiction. Provincial governments make their own decisions about how action plans will be implemented and how critical habitat will be protected on provincial lands. SARA does contain "safety net" provisions that may come into force if a province does not provide adequate protection. However, they have only been used once, for protection of the sage grouse, and only after a successful court challenge (Olive 2015). The reality is that the federal government will not readily intrude into areas of provincial jurisdiction.

Finally, recovery measures are constrained by conflicts with competing land-use objectives. SARA clearly acknowledges that socio-economic factors need to be considered at the action planning stage. However, it is largely mute on how the necessary trade-off decisions are to be made. This has proven to be a point of weakness. Compromises with competing social objectives are often made informally, without transparency or the benefit of a structured decision-making process. In other cases, prescriptions for action are left vague, which avoids conflict but stymies implementation. Because of these shortcomings, there is often little correlation between what is proposed within recovery strategies and what actually happens on the ground.

Land-use conflicts are most acute in the Agricultural South, where recovery strategies must go head-to-head with the livelihoods and legal rights of private landowners. It has long been recognized that species recovery on agricultural lands cannot be achieved by force. Legal arguments aside, heavy-handed efforts to enforce compliance would result in antipathy to recovery programs by landowners, and potentially counterproductive behaviours (Olive and McCune 2017). Furthermore, governments do not have the capacity to monitor the activities of all agricultural producers at all times. Therefore, recovery efforts on private land typically emphasize voluntary collaborative measures that lever the stewardship ethic and positive attitudes toward nature that many farmers and ranchers hold (Henderson et al. 2014).

A contentious aspect of SARA is the reference it makes to compensation (GOC 2002, Sec. 64). On the one hand, it seems unfair to expect individual farmers and ranchers to bear the full economic costs of species protection on behalf of society (Olive 2016). On the other hand, paying compensation sets a dangerous precedent, namely that government will pay private parties, including businesses, for complying with environmental legislation (Smallwood 2003). To date, little money has been paid to farmers as direct compensation for conservation activities. Instead, funding programs, such as the federal Habitat Stewardship Program, generally direct funding to communities and organizations that secure or restore habitat or undertake outreach activities (ECCC 2017a).

In the following sections, we will review the main types of recovery measures included in action plans for species at risk and management plans for other focal species. Working examples are provided in the case studies in Chapter 11. Discussion of monitoring programs is deferred to Chapter 10, as these programs are common to many forms of conservation.

Habitat Protection

Habitat loss and degradation are a concern for most focal species, so habitat protection is among the most widely used conservation measures. The preferred method, from a conservation perspective, is to establish one or more formal protected areas where all resource use is prohibited. But because of land-use constraints, it is rarely possible to establish protected areas tailored to the needs of individual species. Instead, protected areas are usually established through regional initiatives that seek to maximize conservation benefits across multiple species. The needs of focal species must be integrated into these broader planning initiatives (see Chapter 8).

SARA does not demand a prohibition on all human activities within critical habitat, just that the functional role of the habitat be maintained. Therefore, habitat protection often takes the form of limited prohibitions, aimed at curtailing specific types of activities or maintaining specific habitat features. In some cases, blanket prohibitions may be applied, but only for limited periods, such as during the breeding season.

The farther we move from comprehensive protection to narrowly applied prohibitions, the greater the risk that important threats, or the cumulative effect of minor threats, will not be sufficiently accounted for. But higher levels of protection imply less opportunity for resource development. Thus, the determination of how much protection to apply is partly a matter of science, to describe the prohibitions needed to maintain habitat function, and partly a matter of social choice, to identify an acceptable balance among competing objectives.

An important complication in applying habitat protection measures is that the agencies responsible for implementing action plans—primarily provincial fish and wildlife departments—have limited authority over land management. They must collaborate with the agencies and individual landowners that do have this authority. These collaborations can take various forms and are subject to complex power dynamics. In some cases, conservation practitioners simply supply advice or request that the needs of a species be considered (see Box 6.6). In other cases, demands for protection may be harder-edged.

The form of protection applied depends on the type of land management prevalent within a species' range. On public lands subject to resource development, habitat protection is often achieved through industry operating regulations and voluntary best practice guidelines. Land-use planning and protected area initiatives are also important avenues for achieving protection and connectivity. On private land, protection measures include voluntary stewardship, conservation agreements with landowners, and the outright purchase of land (see Chapters 7 and 8).

Box 6.6. Action Plans with Little Action

Because of planning compromises, many action plans lack strong habitat protection measures, even for species that are highly endangered. The action plan for piping plovers in Ontario (EC 2013) provides an example. The three measures that directly address habitat protection are:

- Encourage stewardship activities that conserve or enhance piping plover habitat and increase nesting success
- Incorporate piping plover habitat needs in beach management plans for public and municipal lands
- Continue to provide advice and recommendations to the Lake of the Woods Control Board regarding water level management on Lake of the Woods

The piping plover is an endangered species that ranges across most of southern Canada. Credit: MDF.

It seems unlikely that these non-specific and largely voluntary measures will be sufficient to ensure that the piping plover's habitat needs are secured.

Mitigating Threats

As we saw in Chapter 5, species face a wide range of threats besides habitat loss and degradation. Overharvest, invasive species, pollution, disturbance, and altered ecosystem function are among the most common. In practice, mitigation of such threats is rarely undertaken until there is clarity about what can and should be done, especially if it involves regulation. Consequently, the recovery measures in action plans are often research oriented rather than action oriented.

The action plan for resident killer whales in BC is illustrative (FOC 2017a). A conceptual model is available (Fig. 6.16) that links whale viability with specific threats and these threats with potential **management levers**. Management levers are opportunities for management intervention that directly or indirectly contribute to a desired outcome. These types of models are often referred to as **impact hypothesis diagrams** or influence maps. Despite all that is known about the killer whale—it is among the most well-studied species in Canada—most of the recovery measures in the killer whale action plan involve research rather than management actions.

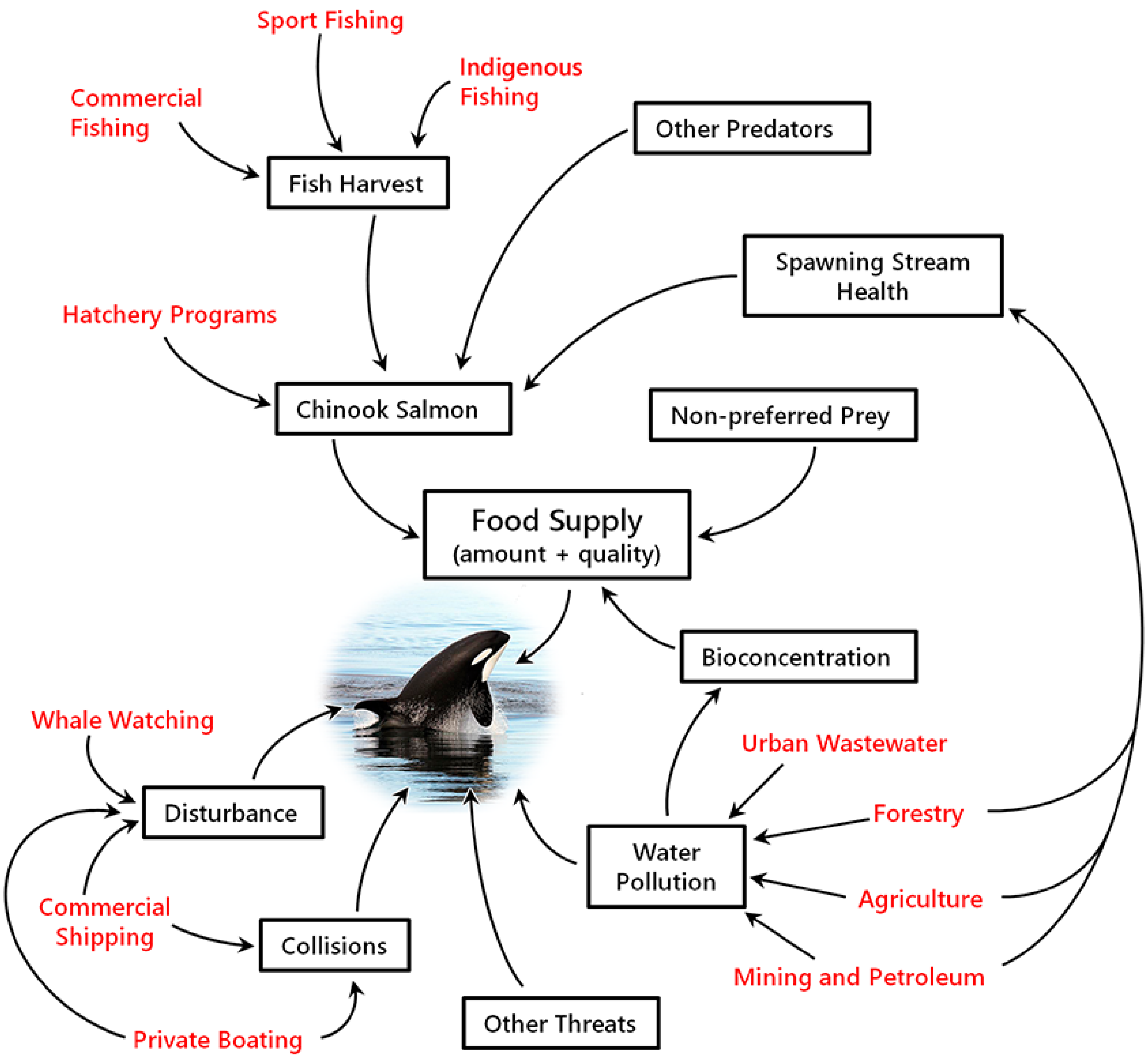


Fig. 6.16. An impact hypothesis diagram illustrating the main threats to resident killer whales along the BC coast, as summarized in the species recovery plan (FOC 2011). Linkages to potential management levers are shown in red. Credit: C. Michel.

Research is needed because management levers are rarely simple on/off switches. Managers need to know how much of an effect is possible, and how much is necessary, and this requires a detailed understanding of the mechanisms involved. There is often pressure to identify minimum thresholds because management actions usually have associated socio-economic costs.

As with habitat protection, the mitigation of specific threats generally requires collaboration with external parties. Returning to our killer whale example, the availability of salmon as a food supply for whales is a critical factor for viability. However, even though Fisheries and Oceans Canada has the legal authority to regulate salmon fishing under both SARA and the *Fisheries Act*, fisheries management is a politically charged process that demands negotiation rather than coercion. Some form of negotiation with external agencies and affected interests is a management reality for most species, particularly for those in the Agricultural South.

The methods used to implement mitigation measures are similar to those used to implement habitat protection. Regulation is a frequently used tool and is applied to industrial practices, pollution, access development, and recreation. Such regulations are often designed to support broad conservation objectives rather than the specific needs of individual species. Regulation is complemented by best practice guidelines, stewardship programs, and education programs.

Augmentation and Reintroduction

In addition to habitat protection and the mitigation of specific threats, species recovery can be enhanced by augmenting the growth of existing populations and by reintroducing populations into abandoned areas of former range (Fraser 2008; IUCN 2013). Successful Canadian reintroduction programs have involved peregrine falcons, wood bison, swift foxes, and many other species (see Case Study 4). Populations may also be introduced into new regions, a topic we will discuss in the context of climate change adaptation in Chapter 9.

Box 6.7. Augmentation Terms

Augmentation: the release of individuals into an existing population of conspecifics. Also referred to as restocking or supplementation.

Introduction: the release of an organism outside of its historical range. In the context of climate change this is often referred to as assisted migration or assisted colonization.

Reintroduction: the release of an organism into a part of its native range from which it has become extirpated in historical times.

Translocation: the intentional release of organisms from one area into another.

Population augmentation can be achieved either by translocating individuals from a donor population or by increasing the number of young produced locally (George et al. 2009). The latter usually involves some form of captive rearing that protects and nurtures young individuals during their most vulnerable period.

Population augmentation is an adjunct rather than a replacement for other recovery methods. Its main role is to reduce the time that a population spends within an extinction vortex. If the population is to become self-sustaining, the root causes of the initial decline must also be addressed. Thus, habitat protection, threat mitigation, and population augmentation should be seen as parts of an integrated package. Tactical modelling of demographics and habitat can provide valuable insight into how these pieces all fit together and how individual projects can be optimized.

Augmentation projects and reintroductions are time-consuming and costly and should not be undertaken without careful consideration of the factors that contribute to a successful outcome. Indeed, surveys suggest that many past projects have been unsuccessful (Godefroid et al. 2011; Cochran-Biederman et al. 2015). Furthermore, because of the costs involved, trade-offs with other conservation opportunities need to be considered (though high-profile reintroductions often draw funding from new sources). The IUCN has developed guidelines designed to ensure that reintroductions are appropriate and effective. These guidelines provide a useful starting point for new projects (IUCN 2013).

An important consideration in augmentation programs is the genetic composition of the individuals being released (Armstrong and Seddon 2008). The source population for breeding stock or translocations should be as similar as possible to the target population so that any existing genetic adaptations will be appropriate for the local environment. But it is also important to maximize genetic variability, as this provides the genetic basis for adapting to environmental change (Jamieson 2011).

In fish, where captive rearing is widely used, it has been shown that a decline in fitness from inbreeding can occur relatively quickly (George et al. 2009). For example, without precautions, the loss of fitness in hatchery-reared salmonids may negate any gains from captive rearing in only four to six generations (Bowlby and Gibson 2011).

The choice of release site is another factor that can influence the success of augmentation projects, particularly for reintroductions (Armstrong and Seddon 2008). Selected sites should feature the habitat conditions associated with maximal population growth across multiple ecological and temporal dimensions. This is another role for habitat modelling. In addition, consideration should be given to future human development trajectories. In some cases, lower quality sites may be preferred if they are better protected than higher-quality areas.

Program outcomes are also influenced by the release methodology. Key factors include the number of individuals released, the age at release, the number of years over which releases are made, and the use of "soft release" techniques that allow individuals to gradually acclimatize to their new surroundings (Holroyd and Bird 2012; Cochran-Biederman et al. 2015). In terms of numbers, the general rule is "more is better," but this is constrained by cost and the health of the donor population.

As for age at release, the aim is to bypass the highly vulnerable stages of early growth. The trade-off is that delayed release increases program costs and may compromise the ability of individuals to exist independently in the wild (Lagios et al. 2015). The optimal choice for each of these factors will differ among species and settings, and some fine tuning through trial and error is generally required (e.g., Mitchell 2011). This calls for an adaptive management approach (see Chapter 10).

Fig. 6.17. Trumpeter swans were reintroduced to Ontario in the 1980s after being extirpated a century earlier. The reintroduction program was led by a government biologist with support from a large team of volunteers. The population now exceeds 1,000 individuals. Credit: M. Nenadov.

Lastly, successful programs require adequate funding, technical expertise, organizational capacity, and continuity (Perez et al. 2012). Stakeholder and public support are also important (Fig. 6.17). In many cases, it is necessary to overcome resistance to actual or perceived constraints on human activities that may accompany a reintroduction. As with the threat mitigation measures we discussed earlier, conservation practitioners usually rely on education, negotiation, and possibly compensation to achieve support.

Trade-Offs

Under SARA, the implementation aspects of recovery planning are supposed to be addressed through action plans. But in practice, these plans only take us part of the way there. Most plans describe what we *might* do, rather than define what we *will* do. This is because the action planning process has lacked the structure and authority needed to grapple with trade-offs among competing objectives (McShane et al. 2011).

We will begin our exploration of trade-offs by examining the concept of conservation triage. Later, in Chapter 10, we will work through a structured approach to decision making that is designed to handle multi-objective planning. Several case studies involving trade-offs are presented in Chapter 11.

Prioritizing Species: Triage

Trade-off decisions are not limited to conflicts between conservation objectives and competing land-use objectives. There can also be trade-offs between competing conservation objectives, which arise as a consequence of capacity constraints. If we cannot undertake all of the conservation projects we would like, then we need to set priorities. These sorts of trade-off decisions are often referred to as **conservation triage** (Bottrill et al. 2008) or **optimum resource allocation** (Joseph et al. 2009), and they have generated considerable controversy within the conservation community. We will examine the debate over conservation triage in detail because it provides useful insights into the complexities of conservation decision making.

In its original battlefield application, triage was a form of medical prioritization used to maximize overall survival in the face of limited resources. Medical care was focused on patients that would benefit most from treatment, and away from patients that were likely to survive without treatment or who were expected to die despite treatment.

In a conservation context, triage has been defined as the efficient allocation of conservation resources to maximize conservation returns under a constrained budget (Bottrill et al. 2008). This is achieved by explicitly accounting for the costs, benefits, and likelihood of success of alternative courses of action. This approach is also referred to as maximizing the return on investment (Murdoch et al. 2007). A prominent example is the prioritization of global conservation opportunities by international organizations like WWF (Bottrill et al. 2008). These organizations funnel their resources to projects expected to achieve the most conservation gain per dollar expended. The concept has also been applied to the prioritization of species, populations, habitats, and mitigation measures, at multiple spatial scales (Carwardine et al. 2008; McDonald-Madden et al. 2008; Joseph et al. 2009; Auerbach et al. 2015; Gerber 2016; Martin et al. 2018). The form of conservation triage that has generated most of the controversy involves the prioritization of species at the national scale.

The proponents of species triage make two key points. First, available conservation resources are usually inadequate relative to conservation need (Gerber 2016). This point is generally not contested. Second, the conventional objective of maintaining all species has the effect of directing a disproportionate share of available conservation resources to the most endangered species because they are in most urgent need of assistance (Wilson et al. 2011). There is an opportunity cost in doing this, as described by Bottrill et al. (2008):

> While resources are spent on actions unlikely to succeed or costly to implement, a whole suite of other assets are likely to receive inadequate investment given a limited budget. The **opportunity cost** of conservation (i.e. what else could be achieved with the same resources or the opportunities that are lost) is rarely reported or evaluated. (p. 650)

For the proponents of triage, what matters most is overall conservation outcomes, not the fate of individual species. The logical course of action is to allocate available resources with maximal efficiency, "such that the marginal rate of increase in viability is equalized across all threatened species" (Possingham et al. 2002, p. 1). Simply put, the aim is to achieve the greatest good for the most species, given the resources available.

The proponents of triage argue that ethical concerns about allowing some species to go extinct do not enter the debate because species losses are the result of inadequate funding, not the efficient allocation of funding (Bottrill et al. 2008). Moreover, they argue that it is better to make the trade-offs between funding and conservation outcomes explicit rather than allowing the public to believe that commitments to maintain all species can actually be achieved at current levels of funding, which they cannot. In this light, triage is simply honest decision making (McCarthy and Possingham 2012).

Opponents of triage do not dispute the need to make optimal decisions. They challenge the assumptions implicit in the triage concept and, by extension, its ability to achieve optimal outcomes (Jachowski and Kesler 2009; Parr et al. 2009; Buckley 2016; Wilson and Law 2016; Vucetich et al. 2017).

First, the opponents of triage argue that a battlefield analogy is not appropriate because conservation resources are generally not moveable assets like bandages or bags of IV fluid. For example, in Canada, much of the funding for conservation programs comes from provincial governments and resource companies. These organizations want to maintain control over their budgets and spend their money locally, rather than optimize the common good. Furthermore, high-profile species, like the polar bear, attract public interest and act as magnets for conservation funding (Small 2011). Any effort to shift resources away from these flagship species on the basis of increased efficiency will be resisted and could well result in a drop in overall funding levels if public interest in conservation wanes. Finally, much of the cost of conservation comes in the form of trade-offs with other social objectives, and these are generally not transferable. For example, the cost of curtailing the salmon harvest in BC to support killer whales is not something that can be redirected to support pine martens in Newfoundland. If conservation resources cannot be readily reallocated, then we are not implementing triage, we are just abandoning difficult cases.

A second shortcoming of the battlefield analogy is that conservation is an ongoing process supported by a *flow* of resources, not a fixed store of resources. Consequently, conservation practitioners are responsible not only for the efficient allocation of resources but also for acquiring resources on an ongoing basis. The opponents of triage suggest that this second responsibility is more important than the first, "to think otherwise may be analogous to arranging deck chairs on a sinking ship in the most efficient manner" (Vucetich et al. 2017, p. 3).

The issue here is not simply a matter of misaligned priorities. The opponents of triage maintain that advocating for triage creates political signals that broadly undermine conservation, outweighing any gains from the efficient allocation of existing resources (Vucetich et al. 2017). Buckley (2016) outlines the problem as follows:

> Advocates of species triage … perceive conservation essentially as an economic optimization problem; and they act as though politics, society, and legislation are a fixed framework, and they are merely tweaking their own operations within that framework. This is incorrect. Advocating triage changes the entire framework. … The current political norm is that extinctions are highly abnormal and regrettable events, that sometimes occur despite our best efforts to avoid them. These norms are embodied in government policy and legislation, agency mandates and budgets, and in the practical politics of social license. … The triage view is that extinctions are normal events within the functioning of a human-dominated planet: a very different position. If it is seen as acceptable to conservationists that one species should become extinct, that signals that it is equally acceptable for other species to become extinct. … In purely pragmatic terms, triage is a poor gambit. (pp. 2–3)

So, an important unintended consequence of triage is that it leaves us rudderless with respect to the objectives of conservation. If we are not trying to prevent the extinction of all species, what exactly are we hoping to accomplish? To suggest that we are optimizing the use of conservation budgets is unsatisfactory because it leaves open the question of how these budgets are determined in the first place. Bereft of the solid ground provided by the objective of maintaining all species, budgeting for species conservation becomes a largely arbitrary process.

This leads us to a third problem with the triage concept, which is ambiguity about what is to be optimized (Wilson and Law 2016; Vucetich et al. 2017). In a battlefield setting, the objective is clear: to maximize the survival of injured soldiers. But this objective cannot be directly extrapolated to species because species, unlike soldiers, are not valued equally. The public places much higher value on charismatic species, and on vertebrates in general, than on simpler life forms (Small 2011). And species that cause us harm are actively targeted for destruction. A different set of priorities exists within the academic sphere, where it has been proposed that evolutionary distinctiveness and contribution to ecosystem functioning are most important (Isaac et al. 2007; Arponen 2012).

There is also a temporal dimension to be considered (Wilson et al. 2011). From a value perspective, proximate and future extinctions are not equivalent. The certainty of loss is higher for species on the precipice of extinction than it is for species not yet endangered, and the consequences of inaction are more immediate and irreversible. The triage approach provides little guidance for how such temporal value determinations should be made, and who should be making them.

The fourth shortcoming of triage involves practical limitations of the optimization process itself. In a battlefield setting, doctors receive continual feedback concerning the outcomes of their triage choices, leading to progressively better decisions. In contrast, species extinction is a long, drawn-out process that is relatively rare in species under active management. Consequently, conservation practitioners must generally rely on mechanistic models or the opinions of species experts, rather than statistical feedback, to predict extinction risk under management. The opponents of triage argue that such approaches do not provide enough confidence for making irreversible decisions involving the fate of species (Parr et al. 2009; Morrison et al. 2016).

Lastly, the flexibility needed for implementing triage at the species level does not exist under SARA. The law could certainly be amended to provide this flexibility. However, it is not at all clear that this could be accomplished without generally weakening the protection that SARA provides to species at risk.

Lessons Learned

Despite its shortcomings, the triage metaphor has been useful in drawing attention to the existence of trade-offs among conservation objectives. And while not offering a fully workable solution, it does point us in the right direction. Given the reality of capacity constraints, we must carefully consider how we allocate the resources available to us to ensure that we obtain the greatest possible conservation benefit.

The reason the triage metaphor stumbles when applied to conservation is that it skips over several important decision-making steps. In a battlefield setting, doctors can proceed straight to making triage decisions because the context, objectives, options, and likely outcomes are all well established. In conservation applications, each of these elements require attention.

A common deficiency of species-level triage proposals is that they lack a clear decision frame. Triage is advanced as an abstract concept, unmoored from the institutional apparatus that governs decision making. We don't know who is making the decisions, the scope of their authority, or what resources are available. Many of the unworkable aspects of triage arise from this lack of institutional context. The practical value of the concept becomes evident when it is applied to specific organizations making decisions about matters they can control.

Another weakness of species-level triage proposals is inadequate decision scoping. A fundamental concern of triage opponents is that triage is working at cross-purposes with broader conservation efforts. This indicates that triage is being scoped too narrowly, without consideration of its role in the broader decision hierarchy. Individual conservation efforts should contribute to the same broad goal, and this requires coordination. Failure of coordination can result in unintended consequences.

The triage debate also illustrates the importance of clarifying objectives. Is the intent of conservation to achieve the most conservation benefit for the most species, as the proponents of triage assert? Or do some species matter more than others, as many Canadians believe? As conservation practitioners, we should recognize that the answers to these questions may lie outside of our expertise, even though they pertain specifically to conservation. We are dealing here with social choices that are determined by values, not science. The implication is that decision making about species requires some form of social input to guide the objectives. This is not something that can be taken for granted.

Finally, effective decision making requires innovative thinking with respect to management alternatives. Shifting resources from one species to another is one solution to inadequate capacity, but it is not the only option. For example, instead of pitting one species against the other, we could manage focal species in groups. Prioritization would then be focused on identifying the management actions that have the greatest collective benefit. This is the idea behind multi-species action plans, which have now been implemented in southern Saskatchewan and several national parks (ECCC 2017b; PC 2016). We will examine one of these multi-species plans in Case Study 4, paying particular attention to the resource allocation process that was used.

In summary, the basic concept of optimizing the allocation of conservation resources is sound, but the triage metaphor itself is too simple to be of much value. Moreover, the concept cannot stand on its own. Optimal resource allocation should be seen as a component of structured decision making, which provides the complete framework needed to integrate policy context, social values, and technical analysis (see Chapter 10).

CHAPTER VII
ECOSYSTEM-LEVEL CONSERVATION

Ecosystem-Level Conservation

Objectives

In the late twentieth century, the scope of conservation broadened from individual species to biodiversity as a whole. This led to the development of ecosystem approaches to conservation, often referred to as “coarse-filter” or “landscape-level” methods (Hunter et al. 1988). The new emphasis on ecosystems was, in part, a necessity born of the impracticality of extending the focal species approach to the myriad species comprising biodiversity. A broad-brush approach was needed. In addition, ecosystems were themselves recognized as a component of biodiversity that merited conservation.

Ecosystems can be defined at multiple scales (e.g., a single bog or an entire forest) and they often blend seamlessly from one to the next (e.g., short grassland to tall grassland). Therefore, ecosystems do not constitute discrete targets for conservation the same way that species do. In practice, we delineate ecosystems using classification systems tailored to specific management applications. In many cases, ecosystem-level conservation measures are applied using administrative boundaries, without regard to the ecosystem types involved.

We will begin this chapter with a discussion of the objectives of ecosystem-level conservation. We will then turn to applied conservation methods and their theoretical foundations. Because most ecosystem-level conservation measures are delivered as part of sector-specific resource management programs, our discussion of conservation methods will be organized by industrial sector. At the end of the chapter, we will examine integrated approaches to land management.

As with the last chapter, we will concentrate on conventional conservation approaches, leaving the accommodation of climate change to Chapter 9. The planning and management of protected areas will also be treated as a separate topic, in Chapter 8.

What Are We Trying to Achieve?

The fundamental goal of conservation is to maintain biodiversity. This implies there is a reference state that we hope to perpetuate or restore. But what exactly is this state?

In general terms, the reference state is usually described as a system under **natural** conditions. The *Oxford English Dictionary* defines "natural" as "not made or caused by humankind." This definition places us in the right vicinity but lacks specificity. Natural landscapes change over time, so there are many versions of "natural" to choose from. It is also unclear how the ecological effects of Indigenous peoples are to be handled (Bjorkman and Vellend 2010). It seems inappropriate to treat ancient hunter-gatherers apart from nature.

What is missing from the dictionary definition of "natural" is context. The "what" of conservation becomes clear once we understand the "why." As recounted in Chapter 2, the modern concept of conservation emerged in the last half of the twentieth century as a societal response to the widespread environmental degradation caused by rapid industrial development. Thus, conservation is fundamentally motivated by a desire to protect biotic systems from anthropogenic threats. And the focus is squarely on the impacts of modern society, not what Indigenous peoples did in the distant past. It follows that the appropriate **ecological reference state** for conservation is the condition of a species or ecosystem as it would be, *today*, in the absence of the anthropogenic disturbances we discussed in Chapter 5. When we encounter or use the term "natural" in applied conservation, we should consider it a shorthand label for this ecological reference state (Fig. 7.1).

Defining the Ecological Reference State

To serve as a management objective for conservation, the ecological reference state must be quantified across all the constituent elements of biodiversity. At the ecosystem level, the relevant components are **composition**, **structure**, and **function** (Wurtzebach and Schultz 2016). Ecosystem composition refers to the variety and abundance of species in a given system. Structure refers to the spatial arrangement of ecosystem components across multiple scales. For example, in forested landscapes, local-scale structure includes the three-dimensional physical structure of a forest stand, and landscape-scale structure includes the mosaic pattern created by the spatial arrangement of individual stands (Shorohova et al. 2023). Function refers to the ecological processes characteristic of living systems, such as succession, nutrient cycling, predator-prey dynamics, and dispersal.

When quantifying the ecological reference state, it is important to recognize that biotic systems are dynamic.

Through natural processes, such as wildfire and animal migration, the elements of a system are in constant flux. Some elements change over the course of the day, some over the course of a year, and others over decades. Consequently, the reference state cannot be captured by a single snapshot in time. Instead, we must characterize the reference state in terms of the **natural range of variability** (NRV; Keane 2009).

Box 7.1. Ecological Integrity

Ecological integrity is a summary measure that describes the condition of an ecosystem relative to its ecological reference state (Wurtzebach and Schultz 2016). The closer a system is to a pristine condition, the higher its integrity. In practice, ecosystem integrity is used mainly as a concept rather than as a practical assessment tool. This is because ecosystems are too complex to be distilled down to a single summary measure. For applied conservation, we measure indicators of individual ecosystem components and express their status relative to the natural state.

In ecosystem applications, NRV is a statistical summary of the mean and variance of biotic elements and processes in a given area under natural conditions (i.e., in the absence of anthropogenic disturbance). The challenge we face in quantifying NRV is that most management areas are no longer natural (Grondin et al. 2018). In some cases, the NRV for such sites can be extrapolated from measurements taken in a nearby protected area. More often, it must be reconstructed. A common approach is to infer the undisturbed state of a system from what is known about its condition prior to industrial development. This is referred to as the **preindustrial baseline** approach.

The preindustrial baseline approach will need to be replaced with a more dynamic approach once ecological transitions from climate change become more extensive (Bergeron et al. 2011). For the time being, it remains adequate for managing most near-term threats, such as habitat loss, fragmentation, overharvesting, pollution, and so forth. We will examine climate-ready approaches in Chapter 9. Note that it is not the ecological reference state concept that needs to be revamped, but how we measure it.

In summary, conservation is motivated by a desire to prevent or reverse harm to biotic systems from the activities of modern human society. The ultimate goal is to have these biotic systems as close as possible to their natural state. This state is described by the NRV of ecosystem composition, structure, and function (i.e., the ecological reference state). In most systems, NRV cannot be measured directly because of anthropogenic changes. It must be extrapolated from elsewhere or reconstructed. We will examine how this is done later in the chapter, in the context of specific management applications. Working examples are also provided in the case studies, in Chapter 11.

To be clear, the ecological reference state defines what we would like to achieve *from the perspective of conservation*. This is not the same as a management target. As we saw in Chapter 6, trade-offs with other land-use objectives also need to be considered, and this generally results in some form of compromise. But it is vital to begin the decision-making process with a clear articulation of what the ultimate objectives of conservation are. This way, the costs and benefits of alternative management approaches can be meaningfully assessed. Moreover, if compromises are necessary, everyone involved will understand exactly what is being given up. This is the essence of robust, transparent decision making.

Rigorous application of the ecological reference state also guards against **shifting baselines** (also known as the "ratchet" effect; Pauly 1995). Shifting baselines occur when conservation objectives are reset each generation based on prevailing conditions, locking in losses that have already occurred (see Case Study 5).

Box 7.2. The Pitfalls of Using Species Richness in Conservation Applications

Species richness is commonly used to describe biodiversity patterns, but it does not provide a useful baseline for applied conservation (Devictor and Robert 2009; Dornelas et al. 2014). The goal of conservation is to maintain biotic systems as close as possible to the natural state, which means retaining the full complement of native species. Species richness is blind to species composition and only provides the raw number of species in a given area. A study of avian species richness in southern Ontario by Desrochers et al. (2011) illustrates the pitfalls of using species richness as a baseline. Progressive conversion of the natural forest in this region to open lands for agriculture and other human uses resulted in the loss of forest specialist species which was masked by an influx of open-habitat species. The integrity of the original ecosystem was markedly compromised, yet species richness was maintained. A focus on richness could even motivate efforts that are counterproductive. As illustrated by the Ontario example, the most efficient way to maintain species richness is through habitat disturbance, rather than by expending the effort needed to maintain sensitive and rare species. This is entirely contrary to the purpose of conservation, which is to protect natural systems from anthropogenic threats.

Institutional Context

To understand how conservation works in practice, we need to understand the institutional framework that both enables and constrains it. This includes the various agencies involved in land management across the country, as well as existing laws, policies, and commitments.

In Chapter 3, we saw that, outside of protected areas, the legal foundations of ecosystem-level conservation are limited and diffuse. Several provinces, but not all, have enacted some sort of legislation supporting sustainable development. In addition, conservation is often incorporated into laws governing specific types of land use, such as forest harvesting. This type of legislation enables conservation by providing high-level direction and authority for management intervention but typically does not compel specific action. A notable exception is federal and provincial legislation governing environmental assessments, which tends to be relatively prescriptive.

Additional support for ecosystem-level conservation is found in government policy. For example, all federal, provincial, and territorial governments have endorsed the *2020 Biodiversity Goals & Targets for Canada*, which features ecosystem approaches (see Box 3.1, Chapter 3). Policy commitments such as these are generally aspirational—they define what we would like to accomplish with respect to conservation, but do not compel action or define accountability for implementation (AGC 2013). In practice, the level of effort put into individual policies reflects the priorities of the political party in power and therefore changes over time.

The implementation of ecosystem-level conservation on public lands is generally overseen by provincial and territorial governments. The federal government's role is mainly supportive, providing funding and helping to align approaches across the country. In addition, the federal government takes an active role in implementing ecosystem-level conservation within national parks, where the maintenance of ecosystem integrity is mandated by law. On private lands, responsibility for conservation rests mainly with individual landowners.

At the provincial level, responsibility for ecosystem-based conservation is divided among ministries that manage specific industrial sectors (Fig. 7.2). Effective integration among sectoral ministries is usually lacking, and this presents an impediment to the delivery of conservation measures. Provincial fish and wildlife departments are also engaged but have little authority over land use. In most provinces, environment ministries are mainly concerned with issues such as water quality and climate change rather than biodiversity conservation.

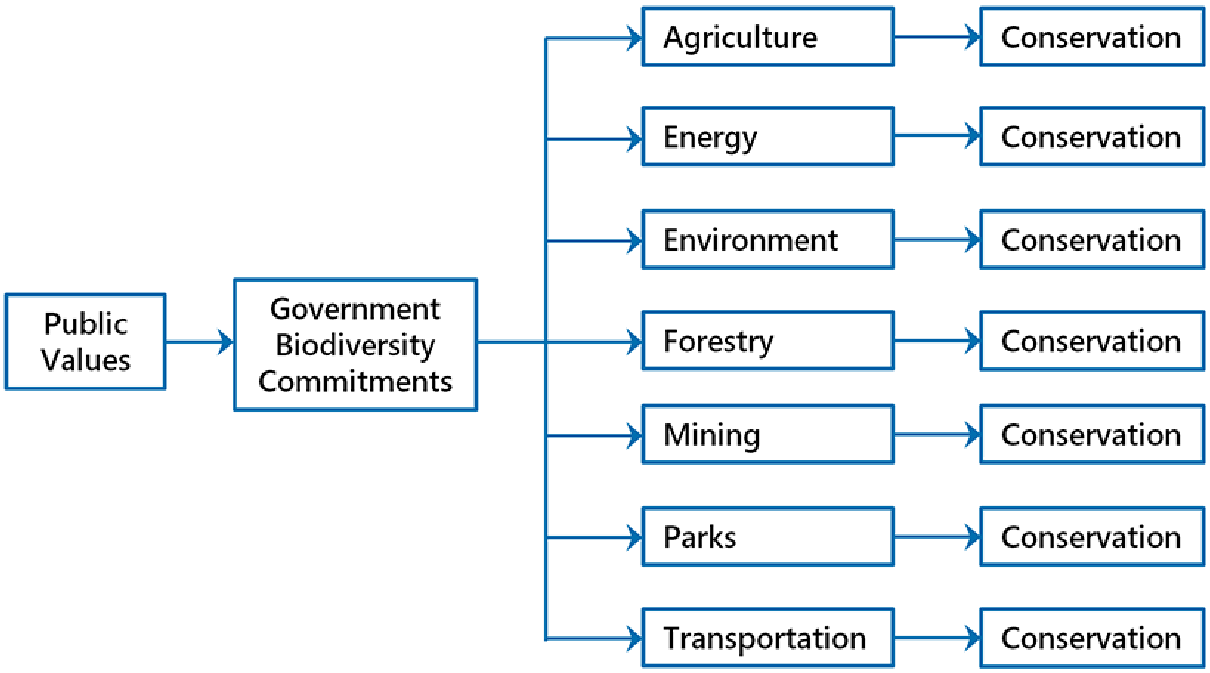


Fig. 7.2. This diagram shows the ministries responsible for implementing ecosystem-level conservation in most provinces. The actual ministry names are variations on what are shown here and some ministries may be combined. Fish and wildlife agencies are typically housed under one of the main ministries.

Because resource companies and farmers are responsible for most landscape disturbances, they have a central role in ecosystem-level conservation. Many conservation measures involve modifications to their operating practices, and they typically bear the cost of implementing these measures. ENGOs, Indigenous communities, and other stakeholders are also important actors, stimulating conservation action and providing a voice for biodiversity in land-use deliberations.

In the following sections, we will review the main approaches used in ecosystem-level conservation. We will examine how each approach works and what it is intended to accomplish. We will also discuss how these approaches are applied in real-world settings, where conservation objectives must be balanced against competing land-use objectives.

Most ecosystem-level conservation approaches are tailored to specific industrial sectors, and this is how we will discuss them. However, integrated approaches also exist, and we will turn to these at the end of the chapter. Discussion of monitoring and adaptive management will be deferred to Chapter 10 because these approaches are not specific to ecosystem-level conservation.

The Forestry Sector

Forestry has the most extensive footprint of all industrial sectors (Fig. 2.9) and it has been the primary target of environmental groups for many decades. As a result, sensitivity to concerns about biodiversity is relatively high, and broad commitments to ecological sustainability have been made.

Forestry sector commitments to ecological sustainability, combined with government downsizing, have resulted in a rather confused mandate for forestry companies. No longer are companies simply in the business of cutting down trees and selling wood products. Today, they are expected to serve as a land manager and achieve a broad range of social and environmental outcomes. Nevertheless, their authority over land use is limited, and the only outcome companies are actually paid for is the production of wood fibre.

A unique and important attribute of forestry companies is that they have the capacity to plan over ecologically meaningful spatial and temporal scales. A typical forest management area is thousands of square kilometres in size and planning extends over a full harvest cycle, implying a timeframe of over 100 years. Also, the registered forestry professionals employed by forestry companies bring a high standard of ecological literacy to the sector. Funding for the planning and implementation of conservation measures is provided mostly by forestry companies, as a cost of doing business, with secondary support from the government (e.g., for high-level planning, management oversight, and research).

There is also a strong institutional foundation for conservation within the forestry sector. There are enabling laws and conservation policies, along with provincial-level regulatory systems that promote sustainable forestry practices and mandate public input. Research centres within academia, government, and industry, along with industry associations, also help to support conservation.

Forestry-based conservation measures are implemented as part of a broader management framework referred to as **sustainable forest management**. The Canadian Council of Forest Ministers (2003) has developed a set of indicators that clarify the values that are to be sustained:

- Biological diversity
- The stability, resilience, and rates of biological production in forest ecosystems
- The quantity and quality of soil and water
- Global ecosystem functions, including the carbon cycle and hydrological cycles
- The flow of economic and social benefits from forests to current and future generations
- Decision making that is fair and effective and recognizes society's responsibility to Indigenous people and forestry-dependent communities

The sustainable forest management framework has helped to raise awareness of non-timber values within the forestry sector and has led to a more balanced treatment of these values in forest management planning (Duinker 2011). It has also led to a tremendous amount of research into sustainable practices. Over the last three decades, our understanding of the ecology of forested systems has greatly advanced. Moreover, research has supported the development of new approaches to harvest planning and operating practices. A key advance has been the development of the natural disturbance model of forest harvesting.

The Natural Disturbance Model

Applied to the working landscape, the objective of maintaining ecological integrity embodies a fundamental contradiction. How can we hope to maintain a landscape that is natural while simultaneously allocating it for industrial development and other uses? This is the ecological equivalent of having our cake and eating it too.

The natural disturbance model (Landres et al. 1999) provides a partial solution based on the insight that disturbance itself is not the problem. Wildfire, storms, insect infestations, grazing, and other disturbances are a natural and necessary part of ecological systems (Fig. 7.3). Over evolutionary time, species have acquired the resilience needed to endure such disturbances without experiencing a long-term decline (Drapeau et al. 2016). Moreover, disturbance is needed to reset succession, generating the young and intermediate age classes that provide ecological niches for many species (Devictor and Robert 2009; Kuuluvainen and Grenfell 2012). The natural disturbance model posits that ecological integrity can be maintained in the face of active land use if human disturbances can be made to emulate natural disturbances (Long 2009; Drapeau et al. 2016).

Fig. 7.3. The Natural Disturbance Model posits that ecological integrity can be maintained if forest harvesting emulates natural disturbances such as fire. Credit: A. Strandberg.

In practice, it is not the actual disturbance processes that are emulated, but the effects these disturbances have on ecosystem structures and patterns (Long 2009). Most applications have involved forested systems where wildfire, insect outbreaks, and windthrow are the dominant forms of natural disturbance (Kuuluvainen and Grenfell 2012). All of these types of disturbance can kill trees and reset successional trajectories, though they do not always do so. There is a great deal of variability in the intensity, location, timing, and extent of individual disturbance events, and this contributes to the complexity of forest ecosystems across multiple scales. Under the natural disturbance model, the aim is to understand this complexity and maintain it through appropriate harvesting practices (Shorohova et al. 2023).

At the scale of individual forest stands, the initial consequence of stand-replacing disturbance is a dramatic change in stand structure. Most notably, the canopy is opened, allowing light and warmth to penetrate to the forest floor. Trees that have been killed often remain standing for many years, contributing structural complexity and a steady supply of dead wood to the forest floor (Fig. 7.4). Given the vagaries of the disturbance process, disturbed areas also tend to have remnant islands of living trees, which further contribute to local structural complexity (Perera et al. 2009).

Fig. 7.4. A lodgepole pine stand regenerating after fire. Credit: R. Schneider.

Changes in stand structure engender changes in species composition (Swanson et al. 2011). In the absence of competition for sunlight and nutrients from mature trees, early successional vegetation is able to flourish. This in turn attracts animal species adapted to early successional habitats. There is also an increase in species specialized for using dead wood (Hannon and Drapeau 2005).

With time, stand composition and structure are remolded by successional processes (Fig. 7.5). The transition of stands to the mature stage is marked by closure of the canopy. Mature stands feature a dense growth of relatively even-aged trees and reduced understory development. The legacy of dead trees slowly diminishes.

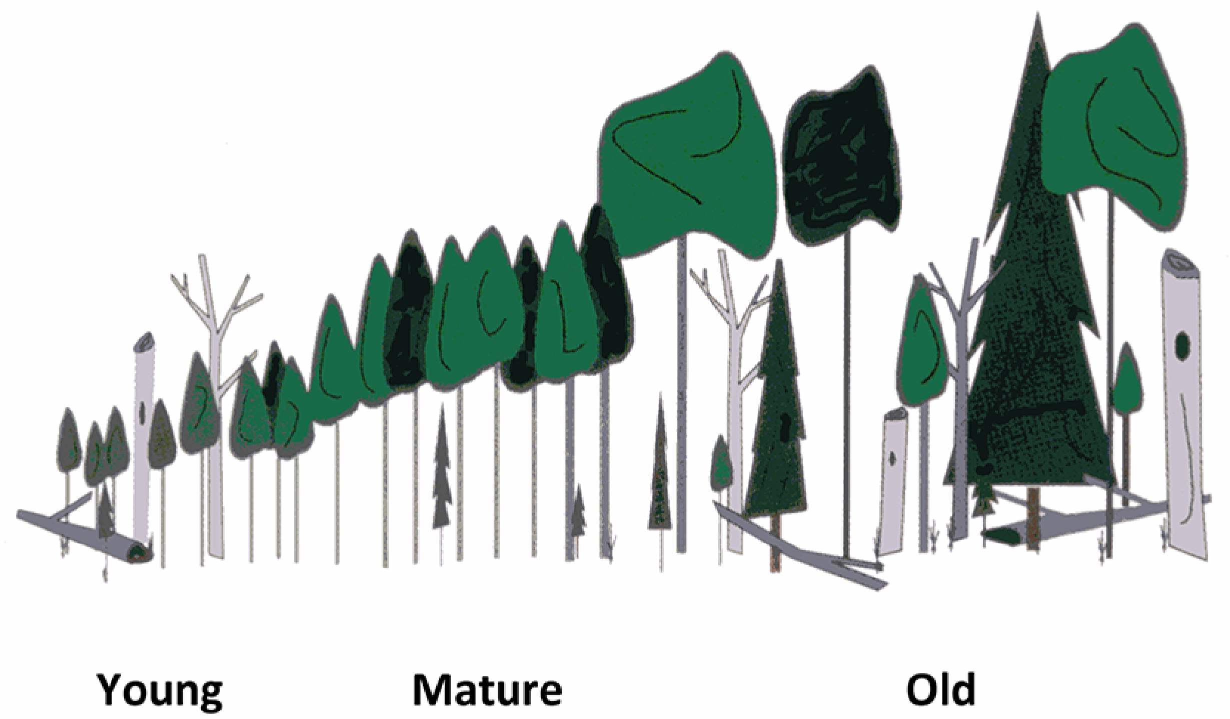


Fig. 7.5. Young stands have an open canopy and contain structural legacies from the stand that was killed. Mature stands have a closed canopy and relatively homogenous structure. Old stands have canopy gaps and complex structure. Graphic by I. Adams.

The transition from mature to old stands is gradual. The key changes include the appearance of canopy gaps from the death of individual trees, release of understory plants, emergence of secondary canopy species, and accumulation of snags and downed logs (Kneeshaw and Gauthier 2003). Relative to younger stages, old stands have trees of many ages and have more large canopy trees, large snags, and large downed logs. Stands with these characteristics are referred to as **old-growth** stands. The age at which this transition occurs varies among species because they mature at different rates. The high level of structural diversity in old-growth stands is associated with high species richness and the presence of many specialist species (Stelfox 1995; Martin et al. 2023).

At the landscape scale, the interplay between natural disturbances, regional climate, physical site conditions, and succession generates complex patterns (Yeboah et al. 2016). Forest patterns are best described as **shifting mosaics** because individual stands are in constant flux as a result of disturbance and succession (Fig. 7.6). Stability manifests at broader scales. For example, the distribution of stand ages across the entire forest may remain relatively constant, even as individual stands turnover. The same applies to the distribution of stand size and to measures of spatial configuration.

Fig. 7.6. An aspen-spruce mixedwood forest illustrating the mosaic pattern that arises from the interplay of natural disturbance, site conditions, and local climate. Credit: R. Schneider.

There are substantial differences in disturbance regimes among regions; therefore, forest patterns exhibit regional differences as well (Shorohova et al. 2023). The western boreal forest often experiences hot and dry conditions during the growing season, predisposing it to large fires (Johnson et al. 1998). Few stands escape burning long enough to reach the old-growth stage. For example, in northeastern Alberta, the mean interval between fires in any given location is approximately 70 years. Consequently, less than 20% of the forest is composed of old-growth stands (Larsen 1997; Al-Pac 2007). These old-growth stands exist as islands within a matrix of young forest.

The rate of burning is lower in the eastern boreal forest because moisture levels are generally higher, and droughts are less frequent (Bouchard et al. 2008). For example, in eastern Quebec, the mean interval between fires is over 200 years. In the absence of harvesting, more than half of the forest is composed of old-growth stands (Belisle et al. 2011). Here it is young regenerating stands that are the islands, in a matrix of old forest. Over time, the old-forest matrix is sculpted by periodic insect infestations, which usually affect older trees (Fig. 7.7). In the eastern boreal forest, insects tend to modify stand structure through selective damage rather than causing stand-level mortality, though exceptions do occur (Bouchard and Pothier 2010).

Fig. 7.7. The spruce budworm is responsible for millions of hectares of forest defoliation each year. Credit: J. Dewey.

Disturbance rates are lowest in the coastal rainforests of BC. Avalanches and windthrow are common sources of tree mortality here, though they usually do not affect large areas (Daniels and Gray 2006). Fires also occur but are very rare. Consequently, only 3% of the landscape is comprised of young and mature stands. The remaining 97% of the forest is in the old-growth stage (Pearson 2010).

Because of these regional differences in forest patterns and composition, the ecological reference state for each planning area must be determined using local data (Shorohova et al. 2023). Funding constraints and practicality limit the number of attributes and the level of detail that can be captured (Table 7.1). Priority is given to attributes relevant to guiding harvesting practices under the natural disturbance model (Patry et al. 2013). The attributes must be amenable to practical, reliable, and cost-effective measurement across large areas (Tear et al. 2005). This tends to exclude many of the functional aspects of ecological integrity, other than disturbance and succession. A working example is presented in Case Study 1.

Table 7.1. Forest attributes commonly used to characterize the ecological reference state under the natural disturbance model.[1]

Category	Attributes
Stand composition	Stand area by type and age class[2]
Stand structure for early, mid, and late successional stages	Disturbance legacy (remaining live trees, standing dead trees, and snags) Amount of coarse woody debris on the forest floor Presence of canopy gaps Uniformity of tree ages within a stand
Landscape Patterns	Stand size distribution Stand shape and spatial arrangement, including the level of fragmentation Spatial distribution of old-growth stands and special features such as riparian zones
Ecological processes	Hydrologic function, including stream flow, turbidity, and connectivity[3]
Human disturbances with no natural analog	Roads, mines, well-sites, hydroelectric dams, etc.

[1]In larger study areas, the attributes shown here may be stratified by regional ecosystem type.

[2]Stand type is based on dominant vegetation and is used as a coarse-filter proxy for overall stand composition.

[3]Disturbance and succession are captured through the stand structure and pattern attributes. Other ecological functions are not commonly measured because of practical constraints.

The ecological reference state is intended to represent the natural state of the system, so any legacies of past industrial use need to be excluded. The most practical way of reconstructing the natural state is through extrapolation from existing undisturbed areas, such as nearby parks, unallocated forest, and those parts of a forestry tenure that are excluded from harvest or are still awaiting their first pass. The main limitation of this approach is that the available undisturbed areas may not be representative of the planning area, either because they are too far away, too small for capturing landscape patterns, or unique in some way. An alternative approach is to use data from a period prior to substantial development, if it is available.

Another complication with reconstructing the ecological reference state from the current landscape is natural variability. As noted earlier, current forest patterns represent a snapshot out of a spectrum of possibilities. Forest age distribution, in particular, is very sensitive to the occurrence of large fires (Cumming et al. 1996). Large fires account for the bulk of the area burned over time, and their sporadic occurrence causes anomalies in the forest age distribution relative to the long-term mean.

Variability in fire occurrence can be handled by reconstructing the long-term, regional history of fires using fire records, tree-ring analysis, charcoal collection from lake sediments, and other techniques (Aakala et al. 2023). Adjustments need to be made to account for anthropogenic fires and the effects of fire suppression, which distort natural patterns. Using these techniques, fire history can often be traced for 200–300 years, permitting estimation of the long-term mean rate of burning and its temporal variance (Belisle et al. 2011). This information can then be used to roughly bound the NRV of the forest age distribution and to derive the natural proportion of old-growth forest.

Variability in stand structure and stand shape are easier to characterize. Most differences are due to fine-scale fire behaviour and subsequent successional processes (Andison and McCleary 2014). Local fire behaviour is in turn largely driven by topography and local weather and fuel conditions (Smyth et al. 2005). So we do not need to reach into the distant past to capture the range of possibilities (Andison and McCleary 2014). The current landscape provides a record of fires that burned under a wide variety of conditions and can provide a reasonable estimate of the NRV of most structural and shape attributes.

The Natural Disturbance Model in Practice

The natural disturbance model has been widely adopted as a tool for guiding forestry operations across Canada, evidenced by its incorporation in government planning manuals in many jurisdictions (Bergeron et al. 2007; Patry et al. 2013). The most notable operational changes that have been implemented include:

- A transition from fixed-size harvest blocks to harvest blocks of variable size
- A transition from square harvest blocks to blocks that emulate natural stand contours
- The on-site retention of snags and downed woody debris during harvest
- The retention of clumps of live trees within harvest blocks (Fig. 7.8)
- The regeneration of mixed species stands in areas where they naturally exist, instead of planting monocultures
- The retention of old-growth stands or old-growth characteristics on the landscape
- A reduction in salvage logging of burned forests

Fig. 7.8. A forest cutblock illustrating irregular contours and the retention of live trees, which are intended to emulate the complex patterns associated with wildfires. Credit: D. Cheyne.

These changes represent a substantial improvement over the earlier system of sustained-yield management. However, conservation efforts remain constrained by a variety of factors (Gauthier et al. 2023). In particular, the calculation of annual harvest rates continues to be based on sustained-yield principles rather than the maintenance of ecological integrity (Burton et al. 2006).

Modelling studies (Armstrong et al. 1999), and experience from implementing the natural disturbance model in the US Pacific Northwest (MacCleery 2008), both indicate that maintaining forest integrity requires a substantial reduction in harvest intensity relative to sustained-yield practices. However, harvest intensity in Canada has only decreased marginally since the introduction of the natural disturbance model (Boyd 2003). With the addition of new forest tenures across the country, the overall area of forest harvested in Canada has actually increased relative to the sustained-yield era.

Additional barriers to implementing the natural disturbance model include:

- **Uneven political support.** Although all provinces have committed to sustainable forest management, through the Canadian Council of Forest Ministers, the level of support and implementation is uneven across the country (Gauthier et al. 2023).
- **Lack of integration.** Sustainable forest management is squarely focused on managing the effects of the forest industry. The ecological effects of other industrial operators in the same area are generally not consid-

ered.

- **Absence of local advocates for biodiversity.** Much of the decision making about conservation measures occurs at the local level, in the context of harvest planning. Although public input has been enhanced under sustainable forest management, most of this input comes from local communities which are dependent on the forest industry for employment. The biocentric views held by a broad segment of Canadian society are often underrepresented.
- **Knowledge gaps.** Despite all the research that has been undertaken in recent decades, the implementation of conservation measures is still constrained by knowledge gaps and uncertainty about the effects of potential management actions.

If we examine the implementation of the natural disturbance model at the level of individual forest attributes, we see that the level of implementation is a function of the associated socio-economic consequences (Drever et al. 2006). For attributes that can readily be emulated, like harvest block shape, managers often seek to maintain the attribute within NRV. But attributes that present trade-offs with other objectives often have management targets that fall significantly short of the conservation ideal (see Case Study 1).

The retention of old-growth forest is a case in point (Gauthier et al. 2023). Efforts to maintain natural amounts of old-growth on the landscape have a direct impact on timber supply and conflict with the conventional practice of truncating the forest age distribution at the optimal harvest age (see Fig. 5.9). Consequently, economic feasibility, rather than NRV, is usually the decisive factor in determining old-growth targets (Patry et al. 2013).

In the western boreal forest, where old-growth stands are relatively rare because of frequent fires, retaining natural amounts of old-growth is feasible as long as the timber supply has not been overallocated (Al-Pac 2007). However, forestry companies must be suitably motivated. In the eastern boreal region, where old-growth comprises a much larger proportion of the forest, natural amounts of old-growth cannot be retained without significant economic disruption. Here, a two-pronged strategy is being explored that uses floating old-growth reserves (see Chapter 8) to extend the age-class distribution of the forest, and silvicultural techniques to emulate old-growth features in younger managed stands (Belisle et al. 2011). Attempts are also being made to implement the triad approach (discussed below) as a way of enabling more old-growth retention while sustaining timber flows. Coastal forests, where disturbance rates are extremely low, present the most difficult case. In these long-lived forests, old-growth retention essentially means permanent protection. Indeed, protected area initiatives, many of them fractious, have been a defining feature of conservation efforts in this region.

Efforts to emulate fire skips (patches of unburned forest) through live tree retention also affect wood supply and are therefore subject to economic constraints. The same applies to maintaining representation of burned forest and insect-killed forest through restrictions on post-disturbance salvage logging. Consequently, management targets for these attributes are typically outside of NRV.

Constraints also exist for emulating the full size range of natural disturbances. In this case, the constraint is public opposition to large harvest blocks rather than economic concerns. When large harvest blocks are used, fewer access roads are needed, which is desirable from a conservation perspective (Carlson and Kurz 2007). But for a broad segment of the public, bigger is simply perceived as worse, reflecting a general aversion to forest harvesting. Moreover, many people mistrust industry and are reluctant to believe that the harvesting of large blocks will actually lead to less harvesting elsewhere. Instead, they perceive the use of large harvest blocks as a slippery slope toward an increased rate of harvest. Because of this opposition, harvest block size is generally capped in the 200–300 ha range, which is orders of magnitude lower than the openings created by large fires (OMNR 2009).

In summary, application of the natural disturbance model should, in principle, ensure that forest attributes remain within NRV despite forest harvesting. However, its implementation generally falls short of the ideal (Gauthier et al. 2023). Our ability to quantify complex forest structures and patterns is limited, as is our ability to emulate those attributes through harvesting. More importantly, the entire process is subject to socio-economic constraints. Consequently, some forest attributes are destined to shift outside of NRV as harvesting proceeds, to the detriment of biodiversity (Grondin et al. 2018).

Zonation

Managing for multiple objectives on the same piece of land unavoidably entails compromise, and as a result, it is often impossible to achieve all objectives satisfactorily. An alternative is to divide the land base into distinct zones that give priority to different management objectives (Anderson et al. 2012). A prime example is the establishment of formal protected areas to support biodiversity, which we will discuss separately in Chapter 8. Here we will examine other forms of zonation commonly used in the Industrial Forest to support conservation.

A form of zonation that has received widespread attention in the conservation literature is the "**triad**" approach (Hunter and Calhoun 1996; Cote et al. 2010). The triad approach adds an intensive management zone to the conventional dichotomy of protected areas and industrial-use areas. The concept was developed in a forestry context, where intensive management means plantations stocked with fast-growing tree species (often exotic or hybrid stock) and silvicultural practices that maximize growth. Additional economic gains are realized from locating the plantations near mills, which reduces road construction and hauling costs (Anderson et al. 2012).

The additional harvest volume generated on plantations is intended to offset production losses elsewhere on the landscape from new protected areas and the robust implementation of the natural disturbance model (Cote et al. 2010). By concentrating timber harvesting in a smaller area there is also a reduction in overall road density (Tittler et al. 2012).

Though the benefits of the triad approach have been demonstrated through modelling studies (Anderson et al. 2012; Tittler et al. 2012), there are few real-world examples. Attempts were made in BC and Alberta but were eventually abandoned because of changes in company ownership and changing economic circumstances (Tittler et al. 2016). The leading example at present is a project in Mauricie, Quebec, which is intended to serve as a pilot for future widespread application of the triad approach in Quebec (Messier et al. 2009).

The limited uptake of the triad approach can be attributed to institutional barriers and to uncertainties that reduce confidence in the expected outcomes. For conservationists, the concern is that, in the absence of legislation, the quid pro quo of increased protection may be less than promised or may not be maintained through time. Low-intensity management zones remain potential targets for future development, whereas plantations cannot easily be restored to natural forest. Furthermore, many conservationists reject the assumption that current harvest levels need to be maintained.

For forestry companies, the concern is that projected gains from intensive management may not be realized. Fire, insect outbreaks, and changing economic conditions may threaten the investments made in establishing plantations and may also lead to future shortages in timber supply (Ward and Erdle 2015). For governments, adopting the triad approach involves major changes in tenure and regulatory systems, which are not readily undertaken.

Another form of zonation involves the designation of **special management zones** (Nitschke and Innes 2004). In this approach, many different types of zones are possible, not just three. For example, special management zones have been used to provide extra protection of caribou calving grounds and to provide buffers around protected areas. They can also be applied to ecological features such as old-growth stands and riparian areas that are difficult to represent in protected areas because they are widely dispersed. In such cases, the sites are usually identified and managed by resource companies as a component of operational planning.

Most provinces employ some form of special management zoning, but there is little consistency in how they are designated or governed. Many special management zones are used to provide ad hoc protection to features of high conservation value where full protection is not feasible. In these cases, protective measures typically involve selective or seasonal prohibitions that are designed to achieve specific conservation objectives. The level of protection generally falls short of the blanket prohibitions of conventional protected areas but is greater than what is provided by generic regulations.

BC's regional land-use planning initiatives of the 1990s and 2000s made extensive use of zonation as a planning tool (Fig. 7.9; FPB 2008). For each planning region, multiple zones and associated management priorities were designated by government planning teams working in collaboration with stakeholders. Zonation was used as a way of proactively tackling land-use conflicts that remained hidden and unresolved under conventional multiple-use management. The number and types of zones varied from region to region, reflecting regional geophysical and ecological differences as well as differences in the priorities of local stakeholders (Nitschke, 2008).

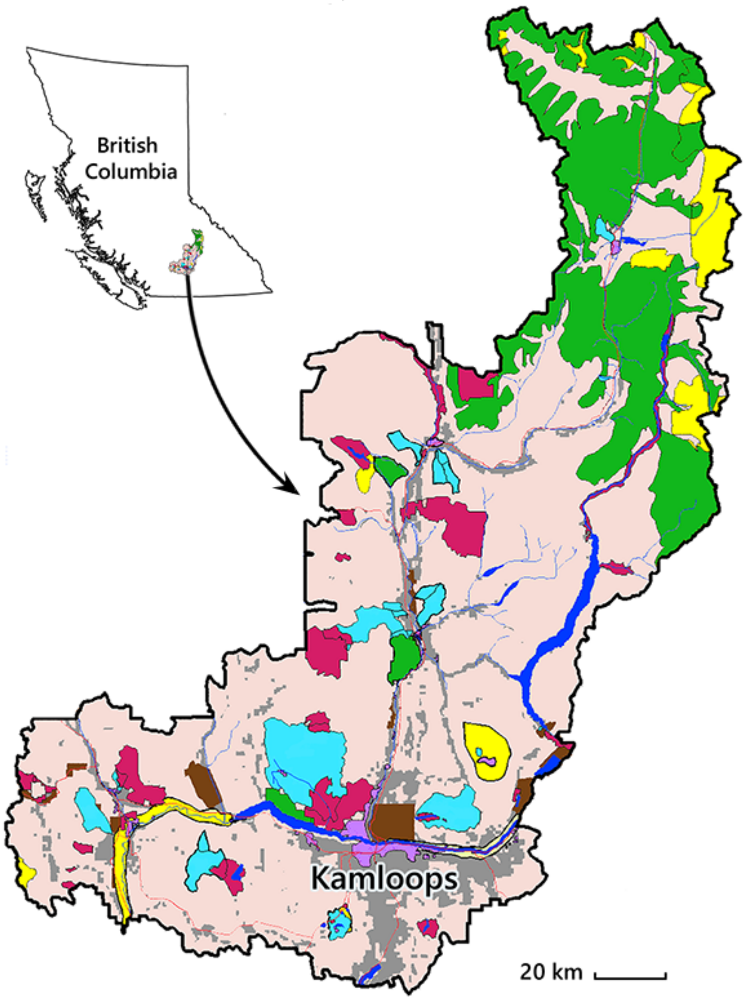


Fig. 7.9. An illustration of zonation, as applied in the Kamloops Land and Resource Management Plan, in southeast BC. SMZ = Special Management Zone. Adapted from KIMC 1995.

The Agricultural Sector

The opportunities for ecosystem-level conservation are limited in the Agricultural South (Fig. 5.5) because most forms of agriculture, as well as urban development, entail permanent ecosystem transformation rather than transient disturbance. This precludes application of the natural disturbance model. Rangelands are an exception because native grasses are often retained to provide reliable forage in the dry regions where rangelands are concentrated.

Another challenge is that few levers exist for compelling farmers to act in the public interest. Most farmlands are privately owned, which provides farmers with strong legal rights over land management. Furthermore, the agricultural sector has considerable political influence at both the federal and provincial levels. Finally, the family farm has a special place in Canadian culture and is strongly supported, even though an increasing number of farms are being sold to corporations and investors. Thus, farming has been largely immune to pressure from environmental groups.

Because of these limitations, conservation measures in the Agricultural South are largely implemented on a voluntary basis, and direct coercion is rarely applied. For example, despite the federal *Fisheries Act* and *Migratory Birds Convention Act*, and various provincial water acts, farmers continue to drain ecologically important wetlands with impunity (Cortus et al. 2011). That said, many farmers have an intrinsic stewardship orientation and will work cooperatively with governments and conservation organizations to achieve specific conservation outcomes (Olive and McCune 2017). These efforts usually target components of the landscape not directly needed for agricultural production.

Environmental sustainability is mentioned in the *Guelph Statement* (FPTMOA 2021), which articulates the current policy vision of the federal and provincial Ministers of Agriculture. However, no conservation framework exists comparable to what has been produced for the forestry sector. Conservation efforts are planned and delivered through collaborative efforts involving federal and provincial agricultural ministries, agricultural producers, and a variety of non-governmental organizations, such as Ducks Unlimited Canada, the Nature Conservancy of Canada, and smaller community groups. Provincial wildlife departments are also involved, though most of their efforts are focused on focal species (which happen to be concentrated in the Agricultural South). Conservation in the agricultural zone is also supported through research at centres across the country.

Ecosystem-level conservation efforts in the Agricultural South include habitat protection (discussed in Chapter 8), the development and use of agricultural best practices, habitat restoration, and the application of the natural disturbance model in rangelands.

Agricultural Best Practices

In areas of active agricultural production, conservation efforts are generally opportunistic rather than target based. The aim is to reduce environmental harm as much as possible through agricultural best practices. These are farming methods that are beneficial to the environment while also being economically viable (Hilliard et al. 2002). The goal of maintaining ecosystems within NRV is only achievable in rangelands and protected areas.

Outreach efforts to farmers are funded and delivered through programs run by governments and various non-governmental organizations. The logistics are challenging. Advancing conservation in the Agricultural South entails interacting with over 200,000 individual farm owners.

The most extensive programs related to the development and promotion of agricultural best practices are run by provincial governments, in collaboration with the federal government. These initiatives tend to use ecosystem services as a conceptual framework (see Chapter 4), and this is reflected in the selection of issues and practices. Best practice guidelines typically include the following categories (Hilliard et al. 2002):

- Nutrient management: precision application of fertilizer and control of manure storage and spreading
- Pesticide management: precision application of pesticides and integrated pest management
- Soil management: conservation tillage and perennial cover programs in high-risk areas
- Riparian management: livestock fencing and seasonally restricted grazing
- Wetland management: natural buffers around wetlands and wetland retention
- Shelterbelts: planting of shelterbelts to control erosion and retain winter moisture (Fig. 7.10)
- Irrigation management: Variable rate irrigation to conserve water and minimize salinization

In most provinces, agricultural best practices are promoted through farm stewardship programs that include government outreach, technical support, and financial incentives. To qualify for technical support and funding, farmers must first complete an Environmental Farm Plan, which identifies environmental risks from farming operations and actions to mitigate these risks (BCARDC 2021). Under this federal-provincial program, farmers can receive funding to implement qualifying best practices on a cost-share basis.

Fig. 7.10. Shelterbelts help to control erosion and retain snow moisture. They also provide added habitat complexity which benefits biodiversity. Credit: R. Schneider.

The best practices included in provincial stewardship programs focus mainly on water and soil, but there are also best practices that relate directly to biodiversity. These practices are designed to maintain as much structural and compositional diversity as possible across the farm landscape (Martens et al. 2013). Examples include:

- Maintain as much perennial cover as possible
- Re-establish and retain natural vegetation, particularly in riparian areas, wetland buffers, shelterbelts, along fence lines, and in remnant patches of forest
- Diversify cropping systems by rotating through different crop types
- Diversify grazing patterns, varying from heavy to light intensity and varying the annual timing of grazing
- Use native grass species in pastures

- Use organic approaches to fertilization and pest control rather than chemical approaches

Programs to support the implementation of biodiversity-oriented best practices are still nascent. An example is the Alternative Land Use Services (ALUS) initiative, which began in Manitoba in 2006 and has since spread to five other provinces (Schmidt et al. 2012). ALUS is a community-led charitable organization whose mission is to help farmers and ranchers build nature-based solutions on their land to sustain agriculture and biodiversity.

ALUS initiatives are implemented on marginal and fragile land on working farms. Participating farmers restore wetlands, reforest native trees and shrubs, plant windbreaks, install riparian buffers, manage sustainable drainage systems, create pollinator habitat, and establish other ecologically beneficial projects on their properties. The intent is to make the most of marginal spaces on farmland, while leaving the better sites for food production. Farmers are responsible for implementing the conservation measures and they receive an annual payment for doing so. Remuneration is based on established cropland leasing rates. Funding partners include a mixture of government, corporate, foundation, ENGO, and community donors. As of 2023, 1,421 farmers had enrolled 15,255 ha in ALUS programs, supported by annual distributions of $4 million (ALUS 2023).

Although initiatives like ALUS are promising, they are constrained by insufficient funding and formidable logistic challenges. The farms enrolled in the ALUS program represent just 0.6% of Canadian farms. It is not that money is lacking; federal and provincial governments spend over $1 billion each year supporting farming in Canada (GOC 2022a). The issue is how farm subsidies are allocated. Kenney et al. (2011) note:

> Over the past two decades, a growing number of studies have highlighted how some government subsidies can serve as a powerful disincentive to sustainability by encouraging overuse and waste of scarce natural resources and placing additional stress on the health of ecosystems. Some subsidies to agriculture encourage agricultural intensification and expansion by either directly tying the level of payments to production levels, or by decreasing the costs of inputs (such as fertilizers and pesticides). … One obstacle to reforming agriculture subsidies is the perception that they predominantly support small family farms and a traditional way of life. The reality, however, is quite different: a 2003 study found that the majority of the subsidies in OECD countries were captured by larger and wealthier producers, which also tend to use more intensive farming practices. (pp. 35–38)

The upshot is that, despite the inviolability of private landowner rights, policy mechanisms exist for promoting conservation on agricultural lands. However, they involve high-level budgets which are politically volatile. In the absence of strong public pressure, which is what transformed the forest industry, change is likely to be slow.

The Natural Disturbance Model in Rangelands

The prospects for biodiversity conservation are much greater in rangelands than in other agricultural areas because grazing is compatible with the natural disturbance model. Moreover, substantial amounts of native prairie remain within rangeland areas, and a large proportion of these lands are still publicly owned (Bailey et al. 2010).

Fig. 7.11. Historically, bison grazing and wildfire were the main sources of natural disturbance on the prairies. To maintain natural ecological patterns in their absence, ranchers can emulate these forms of disturbance with cattle grazing and prescribed burns. Credit: J. Dykinga.

Historically, fire and bison grazing were the dominant forms of disturbance in grassland systems and there was an important interaction between them: fire altered grazing behaviour, and grazing altered the extent and intensity of fires (Fig. 7.11; Fuhlendorf et al. 2006). Together, these processes generated a shifting mosaic of patch types, analogous to what we discussed in forest systems (Anderson 2006).

The application of the natural disturbance model to rangelands involves the use of prescribed fire and cattle grazing to emulate the effects of wildfire and bison grazing, which are now largely absent (Fuhlendorf et al. 2012; Freese et al. 2014). Prescribed fire is already used for brush control, so ranchers have some familiarity with it. Cattle grazing can be used to approximate natural grazing, but the intensity and timing of grazing need to be varied (Bailey et al. 2010). As with forestry, NRV should guide decision making about the timing, intensity, and spatial distribution of managed activities.

In practice, application of the natural disturbance approach to range management lags far behind its application in forestry. Most of the applied research and outreach has occurred in the US (Weir et al. 2013). In Canada, the approach has only been applied in protected areas that have a mandate for maintaining ecological integrity (PC 2008). Outside of parks, range management guidelines often include biodiversity conservation as a desired outcome but make no mention of NRV or the emulation of natural disturbances (Bailey et al. 2010). Practices generally remain focused on sustaining range productivity, implicitly assuming that biodiversity will be maintained by default.

Despite the current lack of implementation, the natural disturbance model still holds promise for range management. Feasibility is actually higher here than in the forestry sector because there are fewer land-use conflicts. The main impediment is funding. As previously noted, it is not realistic to expect ranchers to pay for expensive measures like prescribed fire just because they are in the public interest. New economic instruments will be needed for progress to be made. Also, greater awareness is required within the ranching sector about the natural disturbance approach and its value in conserving biodiversity.

Restoration

Restoration is the process of assisting the recovery of an ecosystem that has been degraded through human activities (SERI 2006). The term “**restoration**” denotes an intent to return a site to its natural state, whereas the related terms “**reclamation**” and “**rehabilitation**” refer to the repair of damage, without necessarily recreating the original ecosystem (Hobbs and Cramer 2008). In this section, we focus on restoration, which is most commonly applied in the agricultural zone. We will discuss reclamation in a later section, in the context of decommissioning mining and oil and gas projects.

Restoration projects are led by a wide variety of organizations, from national groups like Ducks Unlimited and the Nature Conservancy, to local stewardship groups and municipal governments. Provincial and federal governments tend to engage as partners and funders rather than as project leaders. Most initiatives are collaborative and often involve the participation of volunteers (Fig. 7.12). Funding is typically provided by the lead groups or is obtained through government and foundation grants. For example, under the Greencover Canada program, the federal government provided $110 million over five years (2003–2008) to fund the conversion of environmentally sensitive farmland to perennial cover (AAFC 2003).

Fig. 7.12. Restoration programs are typically collaborative initiatives and often depend on support from volunteers. Credit: US National Park Service.

Restoration is expensive and funding is limited. Therefore, it is not a tool that can be applied to all degraded landscapes. Efforts are generally focused on sites with high conservation importance, including ecosystems that have become very rare, habitat for species at risk, and landscapes needed to achieve connectivity objectives. Wetlands that have been drained or degraded through agricultural practices are an example (Tori et al. 2002). The protection and restoration of Canada's wetlands has been a major focus of the North American Waterfowl Management Program since its inception in 1986 (NAWMPC 2012). Remnants of Carolinian forest (Thompson 2011) and native prairie (McLachlan and Knispel 2005) are also common targets for restoration.

Other considerations in site selection include the potential for success (given available funding) and the risk of future degradation. The mandate and priorities of the lead groups and stakeholders are also important. It is no coincidence that Ducks Unlimited works primarily on wetlands.

Another common application of restoration is to return protected areas to a more natural state by reversing changes that occurred prior to their establishment (PC 2008). For example, agricultural fields and orchards accounted for 40% of Point Pelee National Park in southern Ontario in the 1950s, and the site was riddled with hundreds of cottages and access roads. Through a combination of passive and active restoration efforts over several decades, most cottages and roads have been removed and there has been progressive replacement of alien vegetation with native species (McLachlan and Bazely 2003).

In most restoration projects, the emphasis is on restoring the native plant community. It is assumed that native fauna will passively follow once the appropriate vegetation is in place (Hobbs and Cramer 2008). Projects begin by defining the objectives of restoration. The desired endpoint may be complete restoration, but this is usually tempered by practical realities. Restoration efforts are subject to diminishing returns, and the difference in cost between complete restoration and "good enough" may be exponential (Cowan et al. 2010).

It is also necessary to characterize the ecological reference state that will guide restoration efforts. A comparable ecosystem that is still largely intact is often used for this purpose (McLachlan and Knispel 2005). However, new approaches are being contemplated to accommodate global warming (see Chapter 9).

Sometimes, restoration can be achieved through simple passive measures that prevent further degradation but otherwise allow the system to recover on its own through natural processes. In other cases, barriers may exist that interfere with the recovery process. Biotic barriers, such as competition from agronomic species and weeds, are common (McLachlan and Knispel 2005). There can also be abiotic barriers, such as changes in hydrology and increased nutrient levels from fertilizer application (Hobbs and Cramer 2008).

The nature of these barriers, together with capacity constraints, guide the development of restoration strategies. Though generic restoration guidelines are available, such as those produced by Parks Canada (PC 2008), specialized expertise is often required. A significant amount of trial and error learning may also be needed for adapting generic restoration techniques to individual sites (Cabin 2007). In most cases, time and perseverance are critical ingredients of success. Larger projects often require years of sustained effort (Hobbs and Cramer 2008).

Other Industrial Sectors

Mining and oil and gas developments affect less area than forestry and agriculture but often have the highest intensity of impact. Ecological integrity drops to zero in an open-pit mine. Because the disturbances created by these industries have no natural analog, the natural disturbance model is of little utility here. Nor is there an overarching sustainability framework with broad institutional support. Instead, conservation is narrowly applied on a project-by-project basis through environmental impact assessments, operating regulations, and reclamation.

The management philosophy is (1) minimize ecological damage at the time of construction, when most disturbance occurs; (2) minimize on-site and off-site impacts during the operating phase; and (3) restore the site once production has ceased. The working assumption is that the impacts on biodiversity will be transient, given that the disturbances are temporary and affect a relatively small amount of the land base.

A major shortcoming of this management approach is that it does not account for cumulative effects (Franks et al. 2013). The time period between construction and reclamation is long (i.e., decades), so new disturbances are added before old ones have recovered. Moreover, reclaimed sites in forested areas are typically replanted to grass instead of being reforested. Access roads are rarely reclaimed, and abandoned mines and wells are a perennial problem. Consequently, even if individual disturbances are small, the cumulative industrial footprint and its ecological effects can become significant over time (Nitschke 2008). Most jurisdictions in Canada have struggled when it comes to the management of these sorts of cumulative disturbances.

Environmental Impact Assessment and Mitigation

Anthropogenic disturbances with no natural analog, such as open-pit mines, oil wells, hydroelectric dams, roads, and seismic lines are typically managed through the direct mitigation of known environmental effects. Small disturbances and routine developments like access roads and well sites are managed through standardized regulations that typically have only rudimentary provisions for biodiversity conservation. For example, regulations governing the construction of access roads usually include measures to reduce soil erosion, but they fail to address the issues of habitat fragmentation and human access.

The environmental effects of larger projects, such as mines and hydroelectric dams, are addressed through formal environmental impact assessments, as directed by legislation at both the federal and provincial levels. Although each jurisdiction has its own procedures, the basic steps in the assessment process are similar across the country (Connelly 2011). Projects are first screened to determine if a formal assessment is needed. The issues of concern are then identified and assessed.

Environmental assessments are concerned with “**valued ecosystem components**,” which vary from project to project (Connelly 2011). The identification of valued ecosystem components is a government responsibility and usually incorporates the values and priorities of government departments, Indigenous communities, stakeholders, and the public. Biodiversity is just one of many valued components that are examined. Components related to human health (e.g., water and air quality) and ecosystem services (e.g., tourism and Indigenous cultural use) tend to be emphasized.

The effects of a project on valued ecosystem components are assessed through selected indicators. Biodiversity indicators are typically high-profile focal species and there is often little consideration of the impacts of a project on overall biodiversity (e.g., Noble et al. 2016).

Once the indicators have been selected, the effects of the various types of disturbance created by the project are quantified (GOBC 2013). Ideally, these outcomes are expressed in the form of quantitative indicator-specific models that:

- Account for both direct and indirect effects of project-related disturbance on the indicator
- Account for both immediate and long-term effects of project-related disturbance
- Describe project-related indicator changes in the context of natural baseline conditions (i.e., account for any pre-existing deterioration in the indicator)
- Incorporate interactions with other factors known to influence the status of the indicator, including other existing and proposed developments
- Provide an estimate of the uncertainty associated with the predictions

In practice, cumulative effects, interactions among disturbances, and scientific uncertainties tend to be poorly addressed (Duinker and Greig 2006; Jones 2016). In part, this is because project proponents are responsible for the research and analysis, and they view the assessment process as a barrier to be overcome as expeditiously as possible. More generally, individual companies lack the planning capacity, baseline data, and accountability for land management that would contribute to a comprehensive assessment (Fig. 7.13). There is also a general lack of guidance concerning desired regional outcomes, other than the abstract notion of sustainable development. It has long been recognized that a regional approach to assessment is needed to improve project reviews (CCME 2009). This is a topic we will turn to later in the chapter.

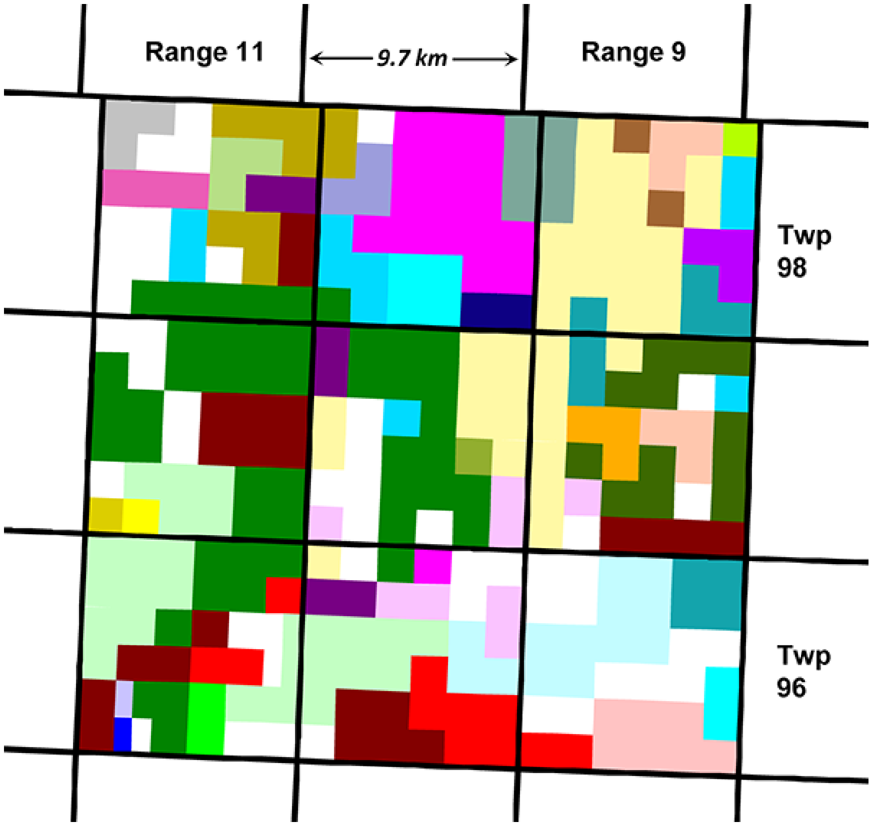


Fig. 7.13. Oil and gas leases are typically one section (1.6 km^2) in size. In this example from northeast Alberta, there are 33 companies operating across an area of nine townships (each company is a different color). These companies are poorly positioned to manage cumulative effects because each has only a small window into the changes that are occurring across the entire landscape.

Assessments must also identify mitigation measures to prevent the potential adverse effects of the project or to reduce them to an acceptable level. Mitigation measures involving project design are particularly important because most land disturbance occurs at the time of initial construction. Efforts can be made to minimize the area disturbed and to avoid disturbing areas of high conservation value. In some cases, mitigation may involve offset measures, in which unavoidable impacts are balanced with restoration efforts in other areas, resulting in no net loss (Noga and Adamowicz 2014). This approach is used in the US but is uncommon in Canada. Environmental impacts that cannot be mitigated are known as residual effects, and they must be described in terms of likelihood, significance, duration, and reversibility.

Based on the proponent's analysis of potential impacts and proposed mitigation measures, the government makes a ruling. In practice, only a small percentage of projects are rejected because the environmental impacts are unacceptable (Fig. 7.14). In most cases, the benefit of the assessment process is in the mitigation measures attached to the approval. To be clear, there is no requirement that all impacts must be mitigated. Approval simply implies that the societal benefits of the project outweigh the environmental risks. This is ultimately a political decision.

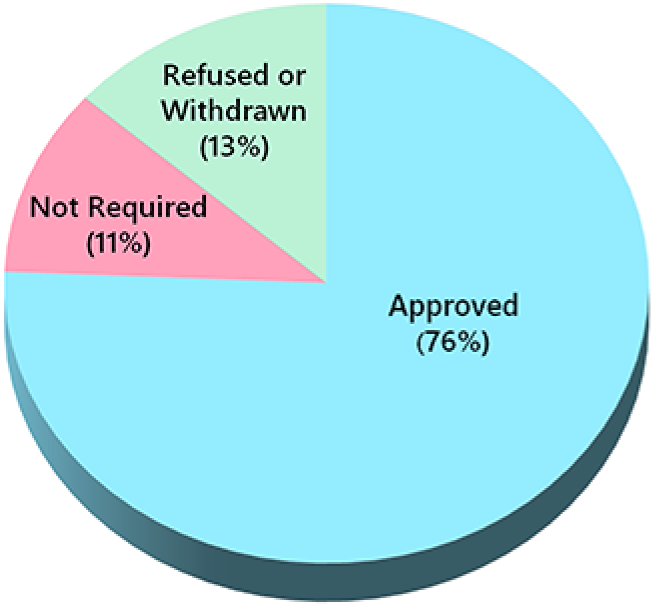


Fig. 7.14. The outcomes of environmental assessments in BC, 1992–2012 (n=156; BCEAO 2012).

Reclamation

When decommissioning industrial projects, resource companies must meet specific reclamation requirements defined in various environmental protection statutes and policies, mainly at the provincial level (Valiela and Baldwin 2007). For smaller projects, like individual well sites, operators are usually required to return the site to an equivalent land capability, rather than restore it to the way it was before disturbance (GOA 2016). In practice, this has generally meant planting sites back to grass (Osko and Glasgow 2010). For larger projects, such as mines, companies are usually required to develop detailed reclamation objectives in consultation with local communities and other stakeholders. Achieving equivalent land capability is again the norm (Fig. 7.15), and an emphasis is typically placed on addressing human health concerns.

Fig. 7.15. Mining operations are required to reclaim sites back to original land capability. This example illustra during operation and after reclamation. Note that the site has been restored to grass, not forest. Credit: Peab

For projects subject to an environmental impact assessment, a formal Closure and Reclamation Plan is normally required (INAC 2007). This plan must demonstrate how the site is to be reclaimed and describe the likely residual risks to human health and the environment. The plan must demonstrate that the knowledge and expertise needed to recreate geochemical, hydrological, and ecological processes, within desired bounds, are available (Johnson and Miyanishi 2008). Companies must also demonstrate that the financial resources for reclamation will be available when required. The overall feasibility of the proposed reclamation program and the seriousness of residual effects are both considered in the project's environmental impact assessment.

The provision of adequate funding for reclamation has been a perennial problem in Canada and elsewhere. Governments are often reluctant to force companies to provide full closure costs up front because this often makes resource projects financially unviable (Cowan et al. 2010). As a result, many companies have been able to circumvent their obligations. There are now an estimated 10,000 abandoned mine sites in Canada, requiring varying degrees of rehabilitation (Castrilli 2010). Abandonment is also a problem in the oil and gas sector. In Alberta, the number of inactive wells awaiting closure has increased from 35,000 in 2000 to 87,000 in 2019, and over 1,700 have been abandoned without reclamation (AER 2020; OWA 2022). Government-industry programs have been established to rehabilitate these abandoned mines and wells, but progress has been slow (Castrilli 2010).

Connectivity

Connectivity is a component of ecosystem integrity that describes how well individuals and populations can move through a landscape (Rudnick et al. 2012). This is an aspect of conservation that applies to all industrial sectors.

The capacity for unhindered movement is needed to support a wide range of biotic processes at multiple scales, including:

- Foraging
- Predator avoidance
- Reproduction
- Dispersal
- Seasonal migration
- Range shifts to accommodate changing environmental conditions

Connectivity may be compromised by physical barriers along important travel corridors. For example, human settlements along river valleys often impede the movement of wildlife, particularly in rugged landscapes where travel alternatives are limited. Dams and roads that bisect watercourses can block the movement of aquatic species. Movement can also be hampered by habitat degradation, which alters movement patterns, slows the rate of movement, and reduces access to certain parts of the landscape (Rudnick et al. 2012). Habitat degradation and fragmentation are most pronounced in the Agricultural South, so this is where the need for management intervention is greatest.

One approach to maintaining connectivity is through broad measures that support wildlife movement across the entire landscape. Some measures focus on maintaining natural ecosystem composition and structure through the application of the natural disturbance model and the proactive control of cumulative effects. Other measures seek to minimize the occurrence of known barriers. For example, road construction can be harmonized between companies to avoid duplication, and water flows can be maintained through proper culvert installation (Robinson et al. 2010). In agricultural regions, farmers can be encouraged to retain riparian habitat and hedge-rows.

Connectivity can also be enhanced through movement corridors (Hilty et al. 2006). In contrast to broad connectivity measures that seek to enhance permeability everywhere, corridors are intended to connect specific parts of the landscape for a specific purpose. A corridor is a type of special management zone in which biotic connectivity is a high management priority.

Corridor projects are implemented across a wide range of scales. Short-range corridors are the most common. They are typically used to facilitate movement across barriers such as major roads and dams and to confer protection to highly used travel routes (Fig. 7.16).

Fig. 7.16. Short-distance wildlife movement corridors are most common. This example illustrates a wildlife overpass on the TransCanada highway in Banff National Park. Credit: WikiPedant.

The largest corridors are continental in scope. For example, the Yellowstone to Yukon initiative aims to establish a 3,200 km interconnected system of wild lands along the Rocky Mountains in Canada and the US (Y2YCI 2014). Another high-profile example is the Two Countries, One Forest initiative which seeks to connect and protect forests of the northern Appalachian/Acadian region from New York to Nova Scotia (Fig. 7.17). These long-range initiatives provide an overarching vision and define regional priorities, but actual implementation usually occurs through smaller local projects, where focused operational planning is more feasible (Beier et al. 2011).

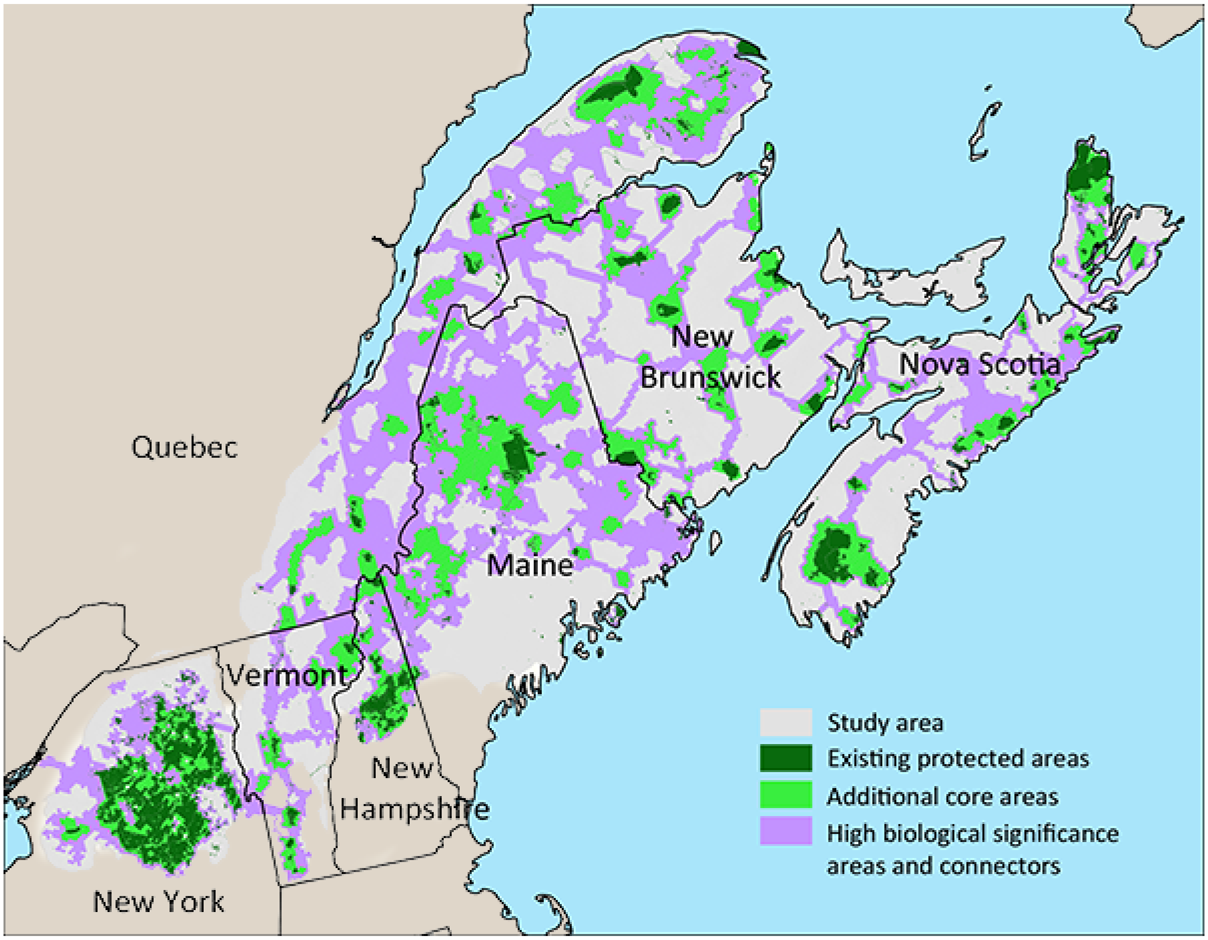


Fig. 7.17. The Two Countries One Forest initiative is intended to provide connectivity between the US Appalachian region and Canada's Acadian region. The proposed design includes core areas and a network of connecting corridors. Source: Reining et al. 2006.

Established guidelines are available that prescribe optimal crossing placement and construction of small-scale wildlife crossings (e.g., Clevenger and Huijser, 2011). Larger initiatives usually require some form of regional connectivity analysis to determine the best corridor design. A key issue is whether the corridor is to benefit a single focal species or many species. A trade-off is involved because species do not all share the same patterns of movement (Perkl et al. 2016). The more a corridor is tailored to the specific needs of one species, the less it will serve the needs of others (Koen et al. 2014a; Dilkina et al. 2016). Corridors for multiple species demand compromise solutions.

Connectivity analysis draws on the landscape connectivity models we discussed in Chapter 6. When the focus is on a single species, assessments are often based on habitat suitability or known animal movement patterns (Rudnick et al. 2012). When designing corridors for multiple species, assessments are more likely to be based on patterns of landscape intactness (Koen et al. 2014a). For regional-scale initiatives, corridor design should be integrated with protected area planning so that potential trade-offs and synergies between corridors and protected areas can be fully assessed (see Chapter 8). The effects of climate change also need to be taken into account (see Chapter 9).

Another aspect of corridor design is determining the appropriate width. Corridor width depends on the target species, length of the corridor, and its intended function. A linear corridor that is 20–40 m wide may be appropriate for a highway overpass (Clevenger and Huijser 2011), but an effective linkage between two distant protected areas may require a special management zone that is many kilometres wide. The appropriate width for long-distance corridors remains a grey area in the scientific literature, so little reliable guidance is available. Moreover, each case is unique.

Lastly, like other conservation measures, corridors are subject to socio-economic constraints. On public lands, trade-offs with other land-use objectives must be considered. On private land, corridor establishment usually requires land purchases and restoration, which can be costly. Furthermore, the creation of long-distance corridors is logistically challenging, in that it requires coordination across multiple industrial sectors and jurisdictions. Consequently, most movement corridors in Canada have been relatively short-range. The creation of long-distance linkages among protected areas remains more a vision than a reality at present. Initiatives like Yellowstone to Yukon are long-term projects that are being implemented piecemeal, through opportunistic local additions.

Invasive Species Control

As we discussed in Chapter 5, alien species disrupt ecological integrity by competing with, preying upon, and displacing native species, thereby altering community structure and function. They also cost the forestry, agricultural, and fishing sectors several billion dollars each year in lost production (EC 2012b). There are well over a thousand alien species now in Canada (CESCC 2022). A small subset with invasive tendencies is responsible for most of the harm. These are mostly plants and insects. Because control programs are very costly, most efforts are focused on these invasive species (Fig. 7.18).

Several federal government departments are involved in invasive species control at the national level. Their efforts are coordinated through the *Invasive Alien Species Strategy for Canada* (EC 2004). Provincial governments (especially forestry and agricultural ministries), academic institutions, non-governmental organizations, and individual landowners (especially farmers) also have a significant role in controlling invasive species.

Fig. 7.18. The Asian long-horned beetle is an invasive species from China with no known natural enemies in Canada. It was first discovered in Ontario in 2003, and because it represents a serious threat to forests, an eradication program was initiated by the Canadian Food Inspection Agency and other partners in 2004. Credit: J. Appleby.

Prevention is widely accepted as the most effective and least expensive means of avoiding or minimizing the risk posed by invasive species (EC 2012b). Border surveillance strategies, including customs inspections, are the first line of defence. These efforts are augmented by regular surveys for high-risk species within the country, often with support from the public. Prevention also involves the use of regulations, best practice guidelines, and public outreach to limit the inadvertent or purposeful movement of invasive species by humans. For example:

- Regulations under the *Canada Shipping Act* prevent ocean-going vessels that enter the Great Lakes from discharging foreign ballast water (GOC 2011a)
- Public education programs in various jurisdictions aim to prevent the spread of invasive species via fishing boats and the transportation of firewood
- Guidelines for reclamation projects and erosion control now prescribe the use of weed-free native species in place of non-native agronomic species

When invasive species are detected, potential responses include eradication, control, and doing nothing. Eradication efforts are typically undertaken when there is a reasonable prospect of success and the cost of action outweighs the cost of inaction. In practice, such cost-benefit analyses usually focus on economic costs rather than ecological costs. For invasive plants, there is a marked decline in success once the infested area increases beyond one hectare (Rejmanek and Pitcairn 2002). Eradication is usually impractical for infestations larger than 1,000 ha. The implication is that the use of eradication as a management tool depends on early detection and rapid response.

If eradication is not possible, then control measures may be undertaken to maintain invasive populations at levels low enough to be tolerable. The decision to institute control programs again hinges mainly on the cost of action (in this case ongoing cost) relative to the cost of inaction. Control programs can be effective but tend to be expensive. For example, the Sea Lamprey Control Program, initiated in 1955, has reduced sea lamprey populations by 90% (EC 2012b). However, the cost of the program is over $21 million per year, shared by Canada and the US. High cost is why control programs are limited to only the most invasive and damaging alien species.

Eradication and control programs employ three main treatment modalities to eliminate target species: mechanical, chemical, and biological (Simberloff et al. 2013). Mechanical approaches are mostly used for plants and include hand-pulling and mowing prior to seed production. This is a safe option that can be used for small infestations of some species, but it is very labour intensive. Chemical control, involving herbicides and pesticides, is generally more effective and can be used to treat larger areas. The trade-off is that chemical agents tend to kill more than just the target species. Chemical control is also expensive when applied to large areas.

Biological control involves the use of living organisms to reduce the reproduction and vigour of an invasive species, dampening its competitive ability. This is a useful approach for large infestations but it also has challenges. Biological control agents are not available for all invasive species and there is a risk of affecting non-target species, potentially irreversibly (Simberloff et al. 2013). For example, the flowerhead weevil, *Rhinocyllus conicus*, was released as a bio-control agent in several Canadian and US sites in the late 1960s to control the alien musk thistle. The weevil has since expanded from the initial release sites and has subsequently caused seed destruction of some sparsely distributed native thistles, potentially threatening their viability (Louda et al. 1997).

Integrated Regional Planning

Each of the ecosystem approaches we have discussed contributes to the maintenance of biodiversity, but these approaches do not constitute a comprehensive system. Much falls through the cracks. Environmental impact assessments help us make decisions about large development projects, but smaller projects, which can transform landscapes over time, are ignored. The natural disturbance model is a useful tool for managing activities like forest harvesting; however, it is ineffective for disturbances that have no natural analog, like roads and oil wells. Zonation separates incompatible land uses but provides no insight into the appropriate level of activity within individual zones. And while mitigation efforts reduce the ecological impact of human activities, they do not address the pace and intensity of development—the ultimate drivers of most environmental decline.

What is needed to address these gaps is an integrated system of planning that manages land use at the regional scale, rather than sector by sector. Regional planning provides the greatest leverage for managing development and ensuring that ecological limits are respected. The catch is that regional planning is also the most complex form of planning and is difficult to apply successfully.

Three approaches to integrated regional planning have been developed by different groups with different mandates (Fig. 7.19). **Ecosystem management** (or ecosystem-based management) is a product of the conservation community that combines a suite of ecological planning concepts into an integrated planning framework. **Strategic environmental assessment** is an approach advanced by the regulatory community for managing cumulative effects. And **regional land-use planning** is a government-led process for making political decisions about conflicting land-use objectives.

Fig. 7.19. Ecosystem management, strategic environmental assessment, and regional land-use planning represent three complementary approaches to integrated planning. The core constituency, focus of interest, and planning boundary of each approach is shown in this diagram. To achieve ecological outcomes, concepts and tools from each approach need to be integrated into a comprehensive system of regional planning.

Each approach contributes concepts and tools necessary for maintaining biodiversity and we will examine each in turn. Unfortunately, a comprehensive system that incorporates all of the essential concepts has yet to be implemented outside of a few specialized cases. In the final section, we discuss why this is so.

Ecosystem Management

Ecosystem management finds its origins in conservation science, emerging as part of the shift to ecosystem-level conservation in the late 1980s. Unlike most of the other approaches we have discussed, it is not a method to solve a particular conservation problem. It is a systems approach to land management built on ecological principles (Grumbine 1994).

We will focus here on the biocentric interpretation of ecosystem management, which predominated when the concept was initially developed (Grumbine 1994; Yaffee 1999). In this interpretation, ecosystem management is advanced as a planning framework with embedded conservation goals. Rather than maximizing the flow of ecosystem benefits to humans, the overriding aim is to maintain the integrity of the ecological systems that provide those benefits (Christensen et al. 1996). In practical terms, this means keeping all ecosystem components within NRV.

A consensus definition of ecosystem management does not exist, but there are several core themes included in most descriptions (Grumbine 1994; Christensen et al. 1996; Long et al. 2015). These themes, listed below, constitute a working definition:

- **Holistic.** Management should be systems based, incorporating the hierarchical structure of ecosystems and the interconnections among elements and scales. Protecting the parts demands protecting the whole.
- **Place-based.** Management should be focused on a specific planning area instead of specific issues or land uses. Planning boundaries should be ecologically meaningful rather than administrative (see Box 7.3).
- **Long-term.** Planning should be conducted over time horizons that are ecologically relevant, and not dictated by political cycles.
- **Scientifically rigorous.** The development and assessment of management options should be based on scientific understanding of ecological processes. The complexity and dynamic nature of ecosystems should be acknowledged.
- **Adaptive.** Current knowledge of ecosystem function should be seen as provisional and incomplete. Because of this uncertainty, management approaches should be viewed as hypotheses to be tested and refined through monitoring programs and applied research.
- **Participatory.** Human activities lie at the root of most conservation problems; therefore, humans must be part of the solution. The development and successful implementation of conservation programs requires input and support from stakeholders and the public.
- **Collaborative.** Given that ecosystems do not respect jurisdictional boundaries, management requires coordination and cooperation among agencies. Collaboration is also needed across scientific disciplines.

The term "ecosystem management" is today widely encountered in resource management. But very few applications adhere to the original framework described above (PM 2009). For example, the forestry sector, which was an early adopter of ecosystem management under the rubric of sustainable forest management, excluded several core elements of the concept. Critically, harvest rates are still based on mill requirements, indicating that the commitment to "nature first" is missing. Furthermore, planning is not truly holistic, in that forest harvesting is the only human disturbance considered, and planning boundaries are not ecologically based.

The most rigorous application of ecosystem management is found in protected areas, particularly in national parks, where maintaining ecological integrity is legally mandated. Strong commitments to ecosystem management have also been made in some regional land-use plans. For example, planning for the Great Bear Rainforest in BC is being conducted using an ecosystem management framework (BCMOF 2012).

Box 7.3. Watersheds or Ecoregions?

A core theme of ecosystem management is that planning boundaries should be ecologically meaningful. Watersheds are commonly recommended for this purpose because they constitute functional systems that can be readily demarcated (Noble et al. 2011; Ball et al. 2013). Moreover, aquatic features in a watershed are intrinsically connected to each other and to the surrounding uplands. Therefore, watersheds provide an ideal foundation for the integrative management of aquatic biodiversity. Watersheds are also an appropriate planning unit for some important ecosystem services, such as drinking water and flood control.

An alternative approach for defining planning boundaries is to use an ecoregion classification scheme like the National Ecological Framework (Fig. 1.3). The ecosystems identified in these types of classifications are determined by physical factors such as climate, landforms, and soils. They match up better with the distributions of terrestrial biodiversity than watershed boundaries, which cut across habitat types. Ecoregion classifications also facilitate the systematic planning of protected area networks, which are intended to provide representation of distinct ecosystem types.

In practice, administrative boundaries are commonly used for planning simply because they facilitate decision making. Crossing jurisdictions, as ecological boundaries do, necessitates collaborative planning, which is difficult to achieve in practice. The upshot is that no single approach is ideal for all applications.

Strategic Environmental Assessment

Strategic environmental assessment was developed by the environmental impact assessment community to improve the management of cumulative effects. It has yet to be widely implemented but is being promoted by the Canadian Council of Ministers of the Environment (CCME 2009).

Cumulative effects are changes in the environment caused by interacting human and natural processes that accumulate over space and time (CCME 2014). Of greatest concern to conservation are industrial disturbances that progressively alter ecosystem composition and structure, and industrial discharges that reduce air and water quality as they accumulate.

The assessment of cumulative effects has been mandatory under the federal *Impact Assessment Act* and its predecessors since 1995. However, as previously noted, cumulative effects cannot be adequately addressed through the keyhole perspective of project-level assessments (Duinker and Greig 2006; Connelly 2011). What is needed is an approach that is regional, government led, and intrinsically forward-looking. This is the role of strategic environmental assessment (CCME 2009). The aim is to help decision makers understand how the choices made today will affect landscapes of the future, providing the foundation for proactive management (see the example in Box 7.4).

Box 7.4. Setting and Achieving Cumulative Effects Limits

A cumulative effects limit, or threshold, is a type of management target. It represents the maximum amount of environmental change that is deemed ecologically and socially acceptable. Setting this target requires the consideration of trade-offs among environmental and resource development objectives and is best handled through a structured decision-making process (Gregory et al. 2012).

The management of cumulative effects is most effective when done proactively. The idea is to take steps early, while flexibility exists, to ensure that the limit is never actually reached (Kennett 2006a). In the absence of proactive management, a cumulative effects limit acts as an on/off switch that is highly disruptive once triggered—like running into a wall at full speed. When this happens, management options are limited, and land users, now facing dire consequences, are likely to lobby aggressively for a reprieve, diminishing the effectiveness of the whole process.

A well-known example of proactive cumulative effects management is the global effort to reduce CO_2 emissions to a level that averts catastrophic climate change (Pacala and Socolow 2004). This example shows that multiple mitigation measures, acting in concert, and early implementation are needed to bend the business-as-usual trajectory toward the desired limit (Fig. 7.20).

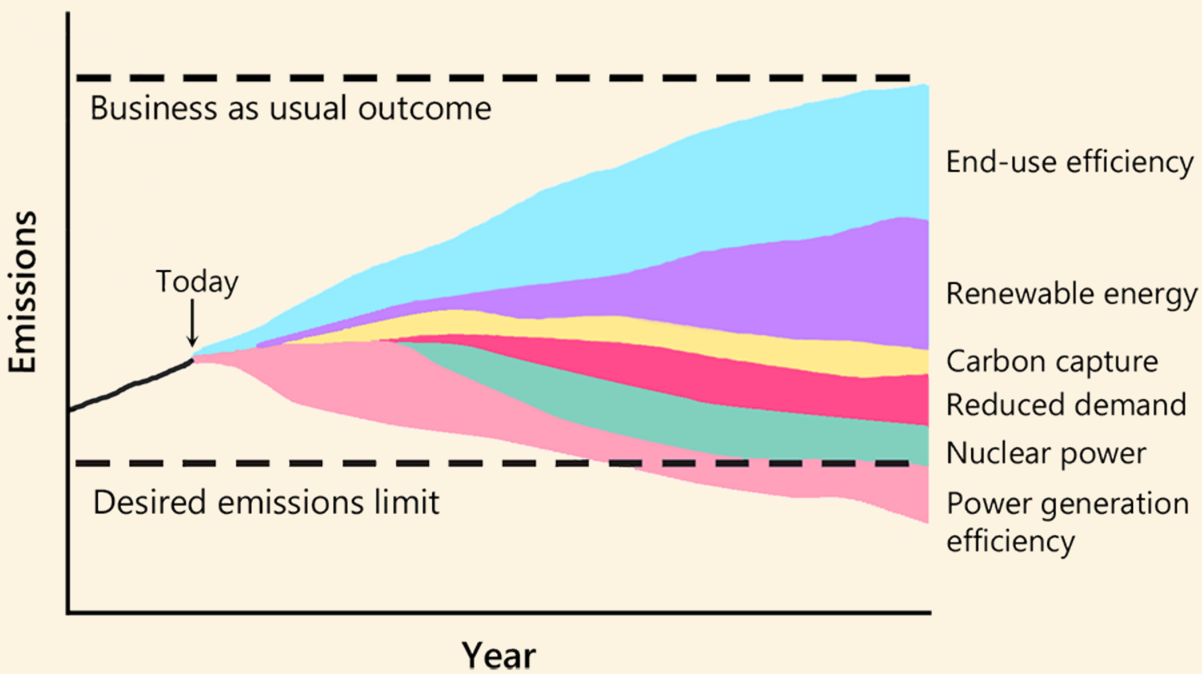


Fig. 7.20. Bending the cumulative effects curve for CO_2 emissions. No single management lever can achieve the desired emissions limit on its own, but the collective contribution of multiple levers, listed at the right, is able to do so. Adapted from Pacala and Socolow 2004.

For biodiversity conservation, the relevant management levers include policy-level curbs on the pace of development (potentially favouring some uses over others), harmonization of infrastructure planning, regulations and incentives for implementing low-impact technologies, and accelerated reclamation, among others. These management approaches are often collectively referred to as integrated landscape management. Experience has shown that industry has considerable ability for innovation once motivated. Case

Study 1 presents an example involving harmonized infrastructure planning that substantially reduced cumulative impacts while also providing cost savings for the companies involved.

There is a natural interplay between the setting of limits and the evolution of management approaches. Limits provide the motivation and focus for identifying mitigative actions. Creative mitigation strategies can, in turn, alter the cost of managing cumulative effects, changing the associated cost-benefit calculations. Thus, over time, planning innovations and technical advances may permit the achievement of cumulative effects limits that were initially considered impractical.

Descriptions of what strategic environmental assessment entails are varied, and there is overlap with other planning approaches (Harriman and Noble 2008). Here we will focus on the analytical tools that are central to its decision support role and which are highly relevant to biodiversity conservation. Similar concepts have been advanced under the labels "**integrated landscape management**" and "**cumulative effects management**."

Strategic assessments incorporate three main inputs: (1) a comprehensive description of the current state of the landscape, including the legacy of past development; (2) an understanding of dynamic ecological processes, including ecological responses to human activities; and (3) a suite of management scenarios. **Scenarios** describe resource development trajectories under alternative management regimes and are chosen to foster learning (Duinker and Greig 2007). It is common to include a reference scenario, representing an unexploited system, as well as a business-as-usual scenario, and a range of other scenarios that emphasize certain values over others or creatively push the boundaries of conventional thinking.

The assessment of outcomes under the selected scenarios usually involves some form of computer modelling (Duinker et al. 2012). Typically, a landscape change model is used to track the state of the landscape under a given development scenario. This information is then fed into sub-models that predict the status of valued ecosystem components, as per the environmental impact assessment paradigm. When focal species are used as indicators, the sub-models are based on the impact hypothesis diagrams we discussed in Chapter 6. Simplified versions are often used because landscape change models can only track a limited set of environmental drivers.

A wide variety of modelling platforms exist. Some, like LANDIS (Scheller et al. 2007) and **ALCES**, provide the full architecture needed for modelling landscape change, which users can adapt to their specific circumstances. Others, like SELES (Fall and Fall 2001), provide users with the flexibility needed to create models of their own design. It is usually the experience and preference of a project's technical team that determines which platform is used.

At the heart of all landscape change models lie a set of rules governing dynamic processes. For example, scenarios for a forested landscape might prescribe different rates of harvest. The model must be able to translate a prescribed rate into a sequence of harvest events, taking into account stand type, stand age, harvest block size, distance from the mill, and other relevant factors. Natural disturbances and vegetation succession must also be modelled, and the interactions between these processes must be accounted for. For example, stands that are burned must be removed from the harvest schedule.

Determining the appropriate level of complexity for landscape modelling is as much an art as it is a science. On the one hand, there are several reasons for adding as much detail as possible. For the model to be seen as legitimate, the simulated industrial practices and natural disturbances need to be realistically portrayed. Also, because the whole point of the exercise is to predict and understand cumulative effects, the model must be able to track small disturbances that add up over time, not just large developments. From an ecological perspective, there is a desire to capture as much detail as possible about changes in ecosystem composition and pattern, to provide a firm foundation for assessing the impacts of development on biodiversity.

The benefits of adding detail must be weighed against the associated costs (Addison et al. 2013). Even though computing power no longer constrains model complexity the way it once did, overall tractability is still a major issue. The time needed for model parameterization, testing, and analysis rises exponentially with model complexity. And in a planning context, time is always of the essence. Furthermore, the more processes and parameters in a model, the more potential points of contention there are. Planning may become bogged down in stakeholder debates over details with little practical relevance, rather than focusing on strategic management issues. Stakeholders and decision makers may also become overwhelmed by complexity, reducing their understanding of the modelling process (Gunn and Noble 2009). Once a model becomes a "black box," that has to be taken on faith, users may become reluctant to trust the results.

In the end, a balance must be struck between realism and tractability (Duinker and Greig 2007). With the advent of powerful computing systems and detailed spatial datasets, the temptation to add detail "because we can" should be avoided. Instead, the level of detail should be determined by the demands of the decision at hand. Furthermore, we must accept that landscape models will never capture all aspects of a system, regardless of how much detail we manage to incorporate (Duinker et al. 2012). They are tools that structure what we know about dynamic processes and allow us to apply this knowledge to improve decision making (Addison et al. 2013).

Regional Land-Use Planning

Whereas strategic environmental assessment is a form of decision *support*, regional land-use planning is a form of decision *making*. The planning process is often conducted by a designated agency, such as a planning board, that operates with a variable degree of autonomy. However, the government is the ultimate decision maker and provides the legal authority for plan implementation.

A regional plan is a roadmap to a desired future. It describes where we are going and how we will get there. Some plans are proactive, outlining a sequence of actions designed to achieve a specific goal, like agricultural development. More commonly, planning occurs in response to conflicts over land use that are not sufficiently addressed by existing regulations (Rayner and Howlett 2009). In this case, the purpose of planning is to devise a solution to a widely perceived problem. This generally entails making trade-off decisions.

Regional plans are best developed using a structured decision-making framework. We will briefly review the main steps in the process, in the context of regional planning, leaving the details to Chapter 10. Like ecosystem management, these steps represent an ideal that is often only partially achieved in practice. A working example of regional planning is provided in Case Study 2.

The first step in regional planning is to frame the decision by identifying the key issues and delineating the planning boundary. Initial framing is usually done by the government and provided to the responsible planning agency through a terms of reference. The planning boundary is a social construct, reflecting the geographic area where the values of interest are located and where the relevant activities take place.

The second step is to clarify the objectives. This requires social dialog because the government does not intrinsically know what the public and key stakeholders want. Environmental objectives are usually included, but not in every case. For example, some regional planning initiatives in BC and Ontario have focused on resolving conflicts between urban expansion and agriculture.

The third step is to identify management options. Regional planners have a variety of management tools available to them (Kennett 2006b). Zonation, which is effective and relatively straightforward to administer, is one of the more common approaches. Other tools control what happens within a given zone. These include rules that limit the intensity of specified activities as well as their spatial distribution and their occurrence over time (see Box 7.4). Operating regulations, economic incentives, and the application of best practice standards may be used to modify activities at a finer scale.

The fourth step is to determine how well each of the management alternatives is likely to perform with respect to the stated objectives. This provides the basis for evidence-based decision making. Computer models are often used to facilitate the evaluation phase, though the level of complexity is quite variable. The highly detailed landscape change models we discussed in the context of cumulative effects management are the exception rather than the rule in conventional land-use planning. When the capacity for modelling does not exist, decision makers rely on expert opinion.

The last step is to choose a preferred management approach. Sometimes the optimal approach is obvious, making broad consensus possible. More often, irreconcilable differences among stakeholders will remain. This does not mean that the planning process has failed. If conducted properly, it will still have clarified the desired outcomes, illuminated fundamental trade-offs, and determined the utility of potential management approaches. Thus informed, the government is in a much better position to make and defend a political decision about how to proceed. Such planning decisions are more likely to be respected if all perspectives have been accounted for and treated fairly, and if the decision-making process has been transparent (Lamont 2006).

Most regional planning initiatives require an ongoing process for plan review and renewal, to keep the plan current as circumstances change. Iterative planning also allows for an assessment of plan's performance against expectations and it supports structured learning about key uncertainties (Gregory et al. 2006).

Barriers to Integration

Ecosystem management, strategic environmental assessment, and regional land-use planning can be thought of as pieces of a puzzle—each contributes something essential to the maintenance of biodiversity, but none is sufficient on its own. Ecosystem management contributes a systems perspective, articulates the ecological foundations of planning, and provides guidance on the appropriate spatial and temporal scales for planning. Strategic environmental assessment provides outcome-oriented planning tools for evidence-based decision making. Regional land-use planning provides the institutional machinery for making political decisions about land use in the face of conflicting objectives, as well as the legal authority needed for plan implementation. Without integration into the government's decision-making process, ecosystem management and strategic environmental assessment have limited potential for influencing regional outcomes.

Over the years, there has been considerable cross-fertilization among the three approaches. However, only limited progress has been made in combining all three approaches into an integrated system of ecosystem-based regional planning (Rayner and Howlett 2009). There are three main reasons for this lack of progress: an unsuitable governance structure, resistance to change, and insufficient political will.

The existing system of land-use governance in Canada is geared toward sectoral decision making rather than integration (Fig. 7.2). Instead of providing a comprehensive vision for land use, existing policies advance sector-specific mandates that often conflict with each other (Kennett 2006b). Moreover, the institutional structure and capacity needed to coordinate land-use decision making are generally lacking (Rayner and Howlett 2009; Kristensen et al. 2013). This situation precludes the systems-based approach embodied by ecosystem management.

Kennett (2006b) has described the governance reforms needed to support integrated regional planning. They begin with a high-level commitment to integration, expressed as a unified vision for land use. In addition, structural changes are needed to vertically integrate decision making from high-level policy down to operational management (Fig. 7.21). Trade-offs should be resolved where it is most effective to do so, and each stage of decision making should lay the groundwork for the next. Legislation is required to ensure accountability and to provide decision-making authority where it is needed.

Another important step is establishing an agency above the sectoral departments that serves as a hub for operational integration—a unified land manager (Fig. 7.21). The role of this land manager is to:

- Develop and periodically revise integrated regional plans
- Provide feedback to elected officials on policy gaps that need to be filled and policy collisions that require resolution
- Provide planning guidance to sectoral departments and ensure that lower-level decisions align with regional plan objectives
- Track land use, monitor regional plan outcomes against stated objectives, and facilitate adaptive learning

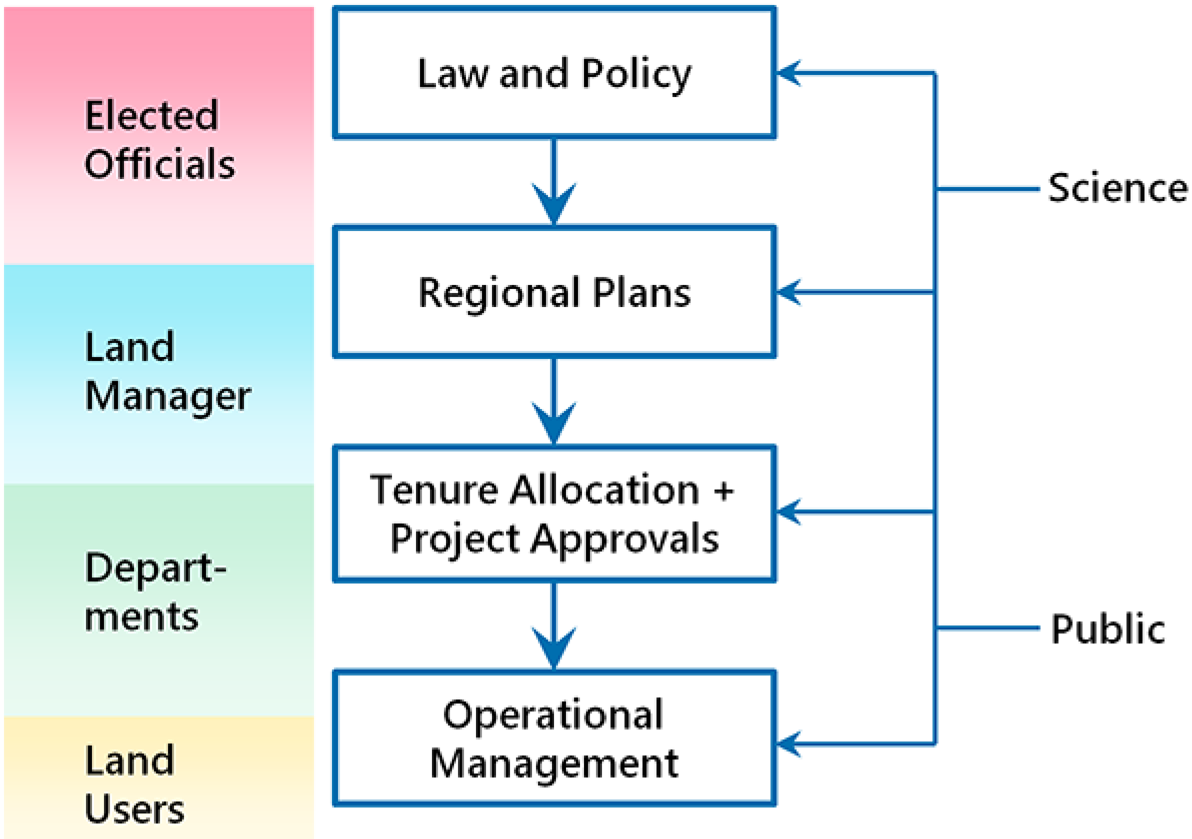


Fig. 7.21. A framework for the vertical integration of planning. Moving down through the hierarchy, the issues become narrowed and the level of detail increases. The column on the left indicates the agency or group responsible for decision making at each stage. Public input and scientific advice enter the process at all stages, to varying degrees.

Little progress on these reforms has been achieved to date, for a variety of reasons. Changes in governance challenge existing power structures, resulting in pushback from those who stand to lose authority, such as sectoral departments (Hodge et al. 2016). Furthermore, a system that promises to balance societal interests is no boon to those who benefit from the existing approach, such as resource companies (Rayner and Howlett 2009). Efforts to achieve balanced outcomes are also constrained by past decisions, particularly those related to project approvals and land tenure allotments. Finally, progress is hampered by gaps in knowledge, inadequate budgets, and the overall complexity of the transition. Case Study 2 illustrates many of these issues.

A commitment to integrated regional planning is ultimately a matter of political will, which has so far been lacking. Governments have been reluctant to undertake institutional change because it requires a significant expenditure of political capital and government resources, yet the political rewards are far from obvious. Though there is strong public support for protection of the environment and biodiversity, this does not automatically translate into strong demand for planning reform. The connection is not readily perceived, and the benefits only accrue over time. Moreover, as noted above, there is substantial resistance to change from many quarters. Governments also realize that integrated planning may lead to contentious debates over land use that ultimately leave all parties unsatisfied. Thus, the status quo tends to be favoured over action.

Given the existing headwinds, progress toward fully integrated planning will no doubt be slow. However, even incremental change will lead to improved conservation outcomes (PM 2009). Conservation practitioners should consider it part of their mandate to advance and promote such improvements.

CHAPTER VIII

PROTECTED AREAS

Protected Areas

Canada's network of parks was built incrementally over the past century. Currently, 12.6% of Canada's terrestrial land base and 9.1% of its marine environment are protected, though not all ecosystem types are equally represented (Fig. 8.1; ECCC 2022). Approximately 35% of the terrestrial parks are under federal jurisdiction, and the rest are managed by provincial and territorial governments or are privately owned. There are several thousand individual parks, but the one hundred largest account for 72% of the total area (ECCC 2022). Two-thirds of the parks are less than 2 km^2.

Fig. 8.1. The distribution of protected areas in Canada in 2022. Source: ECCC 2022.

As recounted in Chapter 2, early efforts at establishing parks were ad hoc and emphasized scenic areas. Since then, systematic planning at the regional and national scales has become more common (Andrew et al. 2014). Federal government efforts have been guided by a long-range national park plan (PC 1997). Provincial and territorial efforts have been more episodic, reflecting waxing and waning political interest, with much variability among jurisdictions (Fig. 8.2).

Many different forms of protection exist, including national and provincial parks, migratory bird sanctuaries, national wildlife areas, wilderness areas, conservation areas, ecological reserves, and other designations specific to individual provinces. These different types of protected areas are associated with different priorities and mandates. In addition to their role in maintaining biodiversity, protected areas serve as ecological benchmarks, support recreation, preserve wilderness, and provide laboratories for research into natural processes (Wiersma 2005).

In this chapter, we will focus on sites where the protection of biodiversity is the primary purpose and industrial activities are specifically prohibited. We will refer to these sites as “protected areas” or “reserves,” and we will use the term “park” when referring to more generic forms of protection.

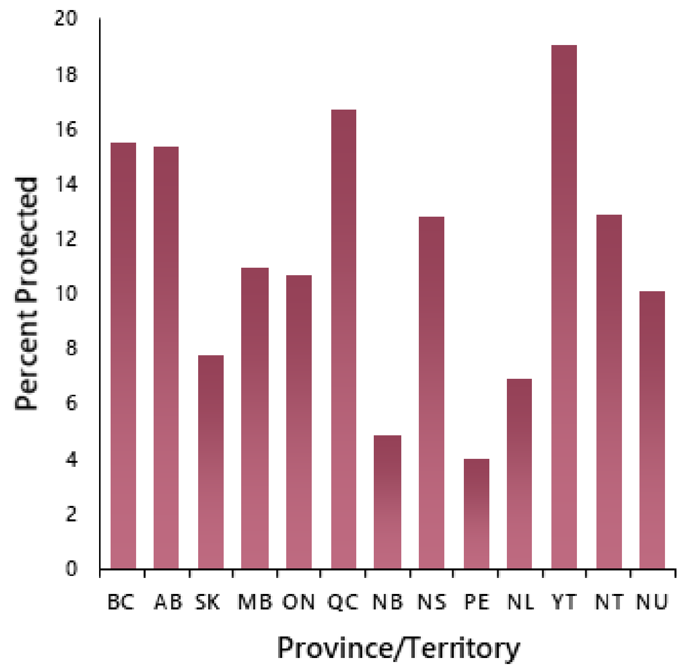


Fig. 8.2. The percentage of Canada’s terrestrial area protected in 2022, by province. Source: ECCC 2022.

Theoretical Foundations

Protected areas have long been a cornerstone of biodiversity conservation. Whereas conservation on the working landscape is characterized by compromise with competing land-use objectives, biodiversity comes first within protected areas. Moreover, there is no uncertainty about whether prescribed protection measures will be implemented or work as intended, given that industrial use is prohibited. The certainty that protected areas provide explains their appeal as a conservation tool. The main limitation of protected areas is that it is rarely possible to set aside enough of the landscape to ensure adequate protection for all species. Therefore, conservation efforts on the working landscape remain vital for maintaining biodiversity, complementing the protection provided by reserves.

Our discussion will focus on reserve design and reserve management. Reserve design remains an important topic because gaps exist in the current network (Andrew et al. 2014). Some regions are underrepresented (Fig. 8.1), and the overall amount of protection falls short of the targets that Canada has committed to. As with the previous two chapters, we will focus on conventional practices and defer discussion of climate change issues to Chapter 9.

Box 8.1. IUCN Protected Area Categories

The IUCN has defined six categories of protected areas. These categories are widely referenced in conservation planning applications (Dudley 2008).

I. Strict nature reserve or wilderness area. Sites where human use is strictly controlled to ensure the protection of natural conditions and biodiversity.

II. National park. Large natural areas that are set aside to protect large-scale ecological processes along with the characteristic biotic components.

III. Natural feature or monument. Generally small sites that protect a specific feature, such as a unique landform or biological community.

IV. Habitat/species management area. Sites that aim to protect particular species or habitats, with management that supports this priority.

V. Protected landscape or seascape. Sites with distinct characteristics that arise from the interaction between people and nature over time.

VI. Protected area with sustainable use of natural resources. Generally large, mostly natural sites where a proportion is under sustainable natural resource management and where low-level, non-industrial resource use is a management objective.

Modern concepts of reserve design trace back to seminal research in the 1960s and 1970s on reserve size and species representation. Biologists working with island ecosystems observed that large islands tended to have more species than small islands. The field of **island biogeography** was developed to explain these findings and to quantify the relationship between island size and species richness (MacArthur and Wilson 1967).

The findings of island biogeography were later extended to reserve design by treating protected areas as islands of natural habitat in a sea of anthropogenic change (Diamond 1975). This gave rise to the general principle that large reserves are preferable to small reserves because they can sustain more species. However, conservation scientists also noted a trade-off. When the amount of land available for protection is limited, a large reserve is not as effective as multiple small reserves in representing all habitat types (Fig. 8.3). This led to a prolonged debate over design priorities, commonly referred to as the “Single Large or Several Small” (**SLOSS**) debate (Simberloff and Abele 1982).

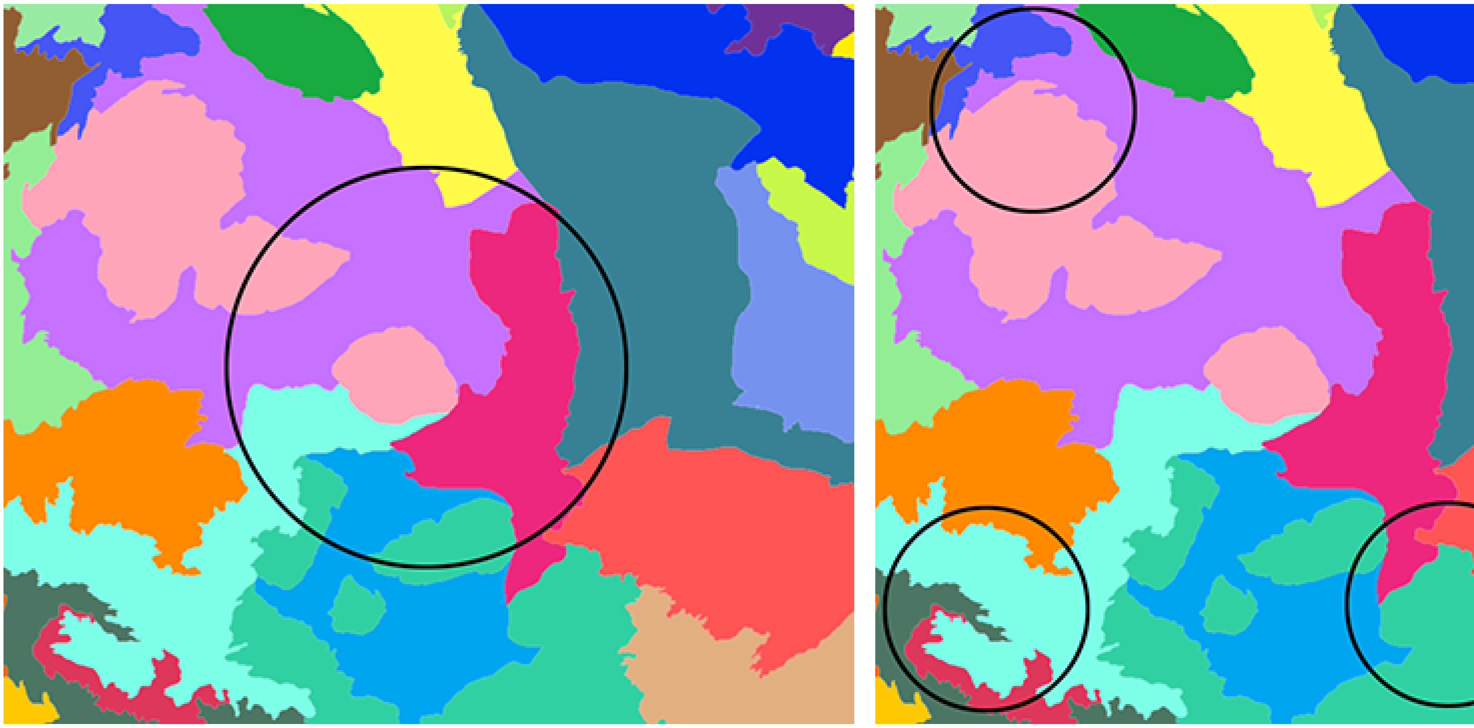


Fig. 8.3. The SLOSS debate centred on the relative merits of a single large reserve (left map) versus several s (right map). Smaller reserves are better able to achieve representation of all habitat types, which can be seer ecodistricts from the National Ecological Framework. However, a single large reserve is more likely to maintai function and retain all ecosystem components. The total area of protection is the same in both maps.

The SLOSS debate was never fully resolved because there is no definitive answer as to which design feature is most important. Size and representation both need to be considered, along with several additional factors. In current practice, the optimal reserve configuration is determined on a case-by-case basis, facilitated by a structured decision-making process. There are two distinct questions that need to be answered. First, how much of the landscape should be protected? Second, what is the optimal spatial configuration of the reserves? We will examine each question in turn.

How Much Is Enough?

Determining the appropriate amount of protection is one of the most challenging aspects of conservation planning (Wiersma and Nudds 2006; Wiersma and Sleep 2018). It is not just a matter of assessing biological need. In real-world applications, trade-offs with competing land-use objectives also have to be considered. The protection of public lands never occurs without this step.

From a strictly biocentric perspective, the preferred solution is to protect the entire landscape. Thus, any protection target below 100% represents a compromise from a biocentric view. The ecological consequences of this compromise depend on how severely the unprotected habitat is ultimately degraded (Fig. 8.4).

Field studies suggest that the effects of habitat loss on biodiversity outcomes are nonlinear (Swift and Hannon 2010; Richmond et al. 2015). Pristine systems are inherently resilient and can withstand small amounts of habitat loss without significant consequences (Fig. 8.4). But as the degree of habitat loss increases, the abundance of sensitive species eventually begins to decline, and certain ecological functions become impaired. Once the majority of the original habitat is lost, a tipping point may be reached where widespread species extirpation occurs.

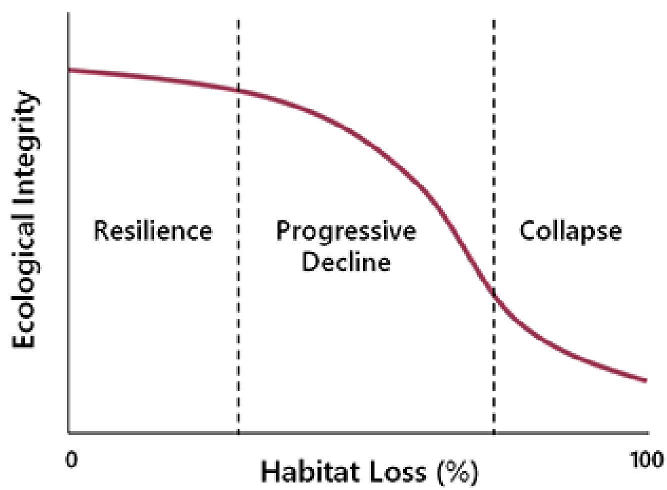


Fig. 8.4. The relationship between habitat loss and ecological integrity is nonlinear. In most applications, the shape of the curve can only be described qualitatively, as shown in this graph.

For planning purposes, it would be useful to know exactly where the tipping point for ecological integrity is, but this information is usually unavailable. Each system is unique, particularly with regard to the types of anthropogenic disturbances present on the working landscape (van der Hoek et al. 2015). It makes a big difference whether the disturbed areas experience complete habitat loss or just a reduction in habitat quality. For most real-world conservation planning, we only have simple qualitative curves, like Fig. 8.4, to work with (Lindenmayer et al. 2005).

Trade-offs with other land-use objectives are best handled through a structured decision-making process. The aim is to compare the benefits and costs of habitat protection across a range of reserve size possibilities using the best available information. From Fig. 8.4, we see that protection provides diminishing returns at high target levels (i.e., within the resilience zone). The inverse relationship holds for costs. Costs are magnified at high levels of protection because little flexibility remains for avoiding areas with high resource potential. The optimal balance between benefits and costs is ultimately a matter of social choice.

Unfortunately, structured, evidence-based approaches are the exception rather than the rule when it comes to setting protection targets. Informal approaches are much more common. Deliberations typically give little or no consideration to the specific biodiversity outcomes that protection is meant to achieve (Svancara et al. 2005). In such cases, it is impossible to determine whether the optimal balance between protection and development has been achieved because the trade-offs are never formally assessed.

Despite the weak foundation of many protection targets, it would be wrong to assume that these targets have been unhelpful. In fact, much has been accomplished. One of the most widely used targets originated with a 1987 report on sustainable development by the World Commission on Environment and Development (WCED 1987). Without any justification beyond the general notion of balancing development and environmental protection, the Commission proposed that the world's existing base of protected areas should be tripled from 4% to 12%.

The 12% target was imported to Canada through the World Wildlife Fund's Endangered Spaces Campaign (1989–2000), which resulted in a doubling in the extent of Canada's protected area network (WWF 2010). The 12% target was important because it provided a bold, yet achievable, goal that inspired and engaged diverse segments of Canadian society and motivated political action.

In 2010, the *UN Convention on Biological Diversity* set a new international protection target of 17% of terrestrial areas and 10% of marine areas (UN 2010). In 2016, the federal, provincial, and territorial deputy ministers responsible for protected areas established a working group committed to achieving the 17% target in Canada.

In 2022, the UN Convention of Biological Diversity again boosted its protection target, this time to 30% by 2030 for both terrestrial and marine areas (UN 2022). The federal government in Canada has likewise begun referencing the 30% target as its protection goal (GOC 2022b). However, behind this headline there has been a shift in the meaning of the term "protection."

Henceforth, protection targets will include "other effective area-based conservation measures." These sites are intended to "achieve long-term and effective conservation of biodiversity, even when *the land is managed for different purposes*" (GOC 2022b). There is as yet no clarity about how the management of these sites will differ from existing approaches, such as sustainable forest management (see Chapter 7). Time will tell whether this approach will lead to a meaningful gain in protection or whether it is simply a method for the spurious inflation of protection targets. Provincial involvement is also an open question at this time.

In summary, protected areas are one of the most effective tools for conserving biodiversity, but they present serious conflicts with other land-use objectives. Compromise is usually necessary. To date, most policy-based protection targets have been far below the level needed to forestall declines in biodiversity (Svancara et al. 2005). That said, protection targets have been trending upwards over time and could continue to do so. Much depends on public perceptions and support for protection. There is also a need for effective and transparent decision making.

Stretch Targets

In recent years, a variety of conservation organizations and academic researchers have been advancing a protection agenda under the tagline "nature needs half" (Noss et al. 2012; Locke 2014; Dinerstein et al. 2017). The 50% target is portrayed as "science based," but strictly speaking, targets are *informed* by science, not *based* on science. From an ecological perspective, there is nothing special about the 50% mark. It is above the point of catastrophic change but well below the point at which ecological decline is detectable (Fig. 8.4).

The selection of 50% as a protection target is basically an expression of risk tolerance—a value judgment on the part of the conservationists involved. Their intent is to reframe the public discourse about protected areas by advancing the possibility of protecting much more than 12% or 17% of the landscape, where opportunities exist. Their general point is that we should not be fixated on averting catastrophic ecological loss—we should aspire to maintain a high level of ecological integrity. The appeal of the 50% target is that it is simple and clean, and the tagline "nature needs half" is compelling. As such, it is well suited for its role in conservation advocacy.

Systematic Conservation Planning

The design of reserve networks has been of central interest to conservation biologists from the outset. The core elements of effective design are now well established under the rubric of systematic conservation planning (Margules and Pressey 2000; Kukkala and Moilanen 2013). Systematic conservation planning is a form of structured decision making adapted to the identification of optimal reserve designs. We will examine each step in turn, leaving the refinements needed to accommodate climate change to Chapter 9. A working example is provided in Case Study 6.

Decision Framing

Most protected area initiatives on public lands involve some form of government process. The government has ultimate responsibility for land management and holds the authority for designating new reserves.

As the final decision maker, the government is normally responsible for framing the decision. The planning area must be identified, the purpose clarified, the core stakeholders selected, the level of public consultation described, and the amount of financial and technical support established. Fixed constraints, such as exclusion areas, also need to be defined. External parties will often seek to influence the framing process, particularly as it relates to objectives, transparency, and stakeholder involvement in decision making.

Parks Canada is the lead agency for protected area planning at the federal level. At the provincial level, protected area initiatives are often part of broad land-use planning programs that involve multiple ministries. These programs are typically overseen by the lead agency responsible for land management, rather than the provincial parks department.

Protected area planning is also undertaken independently by conservation organizations and conservation scientists. These initiatives help to identify conservation priorities and are important in setting the political agenda. However, these organizations lack the authority to establish protected areas on public lands. Nor can they provide an appropriate forum for social decision making. To be implemented, their proposals must eventually be funnelled through a government-run process. Unfortunately, many proposals languish because no linkage is ever made (Knight et al. 2008).

Protected area initiatives on private lands follow a different process. Efforts are typically led by conservation organizations, rather than the government, and they entail direct negotiations between conservationists and landowners. We will discuss the particulars of conservation planning on private lands later in the chapter, in the context of habitat protection in the Agricultural South.

Selecting Biodiversity Surrogates

If a reserve network is to support all elements of biodiversity, then all elements need to be represented in the system. The problem is, we know very little about the distribution and habitat needs of most species. Moreover, there are far too many species to work with individually. Therefore, most conservation planning initiatives achieve representation of biodiversity indirectly, using surrogates (Sarkar et al. 2005). The idea is to select a manageable set of well-described biodiversity components and have them stand in for the other elements.

A common method of selecting biodiversity surrogates is the **coarse-filter** approach (Hunter et al. 1988). Rather than working with individual species, the coarse-filter approach seeks to identify and represent the major ecosystem types within a planning region. The assumption is that, by representing all major ecosystem types, the habitats of most species will be represented as well. Moreover, ecosystems are a component of biodiversity that merits representation in its own right (Margules et al. 2002).

In Canada, several ecosystem classifications have been developed to support coarse-filter planning. A prominent example is the *National Ecological Framework* (Marshall et al. 1999), which is a hierarchical ecosystem classification system based on landforms, soils, climate, and dominant vegetation, among other attributes (Fig. 1.3). Several provinces have developed their own systems of ecological classification at finer scales, harmonized to a greater or lesser degree with the national system. These ecosystem classification schemes have served as the foundation for many government-led protected area planning initiatives in recent years, at both the federal and provincial levels (Lemieux et al. 2010).

Another approach to achieving broad representation is to use **umbrella species** as biodiversity surrogates (Fig. 8.5). These species have large area requirements; therefore, providing adequate protection for them ensures that many other species are protected as well. However, this is not a systematic approach and it does not ensure comprehensive representation in the way the coarse-filter approach does (Higgins et al. 2004). The benefit of using umbrella species is that they are usually well known and can generate public support for protection. For example, linking habitat protection to the needs of grizzly bears was critical to the establishment of the Great Bear Rainforest in BC.

Fig. 8.5. Species with large home ranges, such as the grizzly bear, are sometimes used as biodiversity surrogates in conservation planning initiatives. Such species often have high public appeal and tend to boost support for protection. However, this approach does not provide systematic representation of all habitat types. Credit: J. Frank.

A limitation of using biodiversity surrogates for conservation planning is that species with unique habitat requirements may be overlooked (Mac Nally et al. 2002; Stewart et al. 2018). These species are often referred to as **fine-filter species**, indicating that they have slipped through the mesh of the metaphorical coarse filter and require specialized attention. The habitat needs of these species must be addressed on an individual basis (i.e., a fine-filter approach) rather than through a surrogate approach (Hunter et al. 1988). In practice, this is often difficult to do because many fine-filter species are rare and difficult to study. In many cases, information about their habitat needs is rudimentary.

Most conservation planning initiatives also include species of social importance. These are not true fine-filter species, in that they are not distinguished by unique habitat requirements, but they tend to be treated as such in practice. A better term for them is **focal species**. In principle, the inclusion of focal species is redundant, since their habitats should be represented through the coarse-filter approach. Nevertheless, stakeholders often want to have species of special significance directly represented to provide assurance that their habitat needs are indeed being addressed.

The choices we make about which surrogates to use as coarse- and fine-filter elements provide an operational definition of biodiversity (Sarkar et al. 2006). There are no generic standards to guide these choices, and in practice much depends on data availability and the knowledge and biases of the individuals involved in the planning processes (see Case Study 6). This is one reason why protected area planning initiatives differ so much from one to the next.

Design Objectives

The purpose of the reserve design process is to determine the optimal configuration of reserves for a given protection target. Several design attributes contribute to the effectiveness of reserves and must be taken into account. The optimal configuration is that which scores highest in terms of overall effectiveness. What makes this determination challenging is that trade-offs exist among many of the design attributes.

A fundamental design objective is to represent all components of biodiversity within the network. Protected areas only benefit the biodiversity elements that are included in the system. Representation objectives are usually expressed as minimum area targets for each of the selected biodiversity surrogates and fine-filter elements.

Consideration must also be given to design attributes related to ecological function (Rothley 2006). Reserves are not like stamp collections, where representation is an end in itself. Reserves exist to maintain biodiversity, and so individual reserves must be able to provide effective protection for the species they hold (Rouget et al. 2003).

One of the main functional attributes is the size of individual reserves. Ideally, reserves should be large enough to maintain natural ecological processes, including the interplay between natural disturbance and succession (Leroux et al. 2007). In forested areas where large fires are the major disturbance agent, the minimum reserve size for maintaining dynamic processes has been estimated to be over 4,000 km^2 (Leroux et al. 2007). Redundancy is also desirable because large natural disturbances can result in catastrophic losses within individual reserves.

For focal species, a reserve should ideally be large enough to independently support a viable population. For large mammals, this may require a reserve size of several thousand square kilometres (Gurd et al. 2001).

Reserves should also have a large interior to edge ratio, to minimize the impact of anthropogenic disturbances occurring in the surrounding working landscape (Woodroffe and Ginsberg 1998). Edge effects are of greatest concern in small reserves because the amount of edge habitat becomes proportionately greater as reserve size decreases (Fig. 8.6). Shape is also important, in that round reserves have proportionately less edge habitat than elongated reserves of the same size.

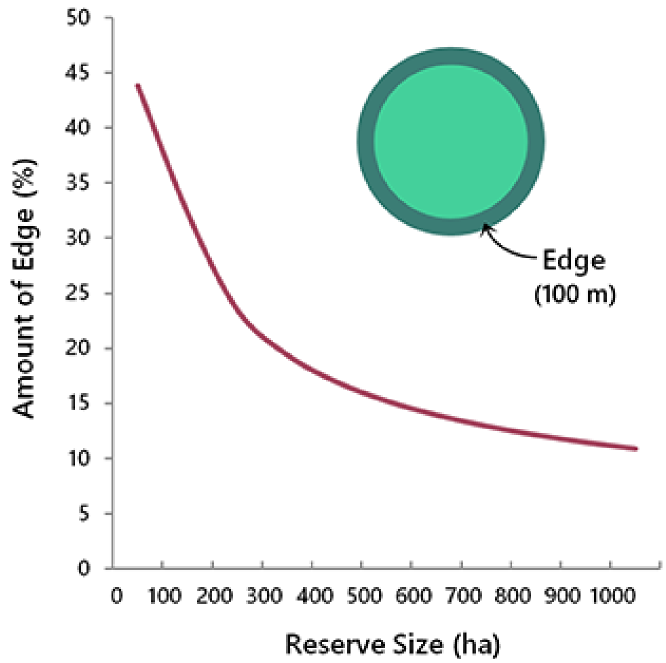


Fig. 8.6. Edge effects are a major concern for small reserves. This graph illustrates the percentage of a reserve affected by edge effects as a function of reserve size, given edge effects that extend 100 m into the reserve.

Another important factor in reserve design is connectivity. Reserve configurations that minimize the distance between reserves, incorporate natural movement corridors, maintain hydrological connectivity, and build on existing reserves are desirable (Rouget et al. 2003). Such designs support gene flow, facilitate species recovery after a major disturbance, and help species accommodate to climatic change.

The level of pre-existing disturbance is also a design consideration. Reserve designs that achieve representation objectives while avoiding disturbed areas as much as possible are generally favoured because ecological integrity is assured and comes at no cost.

Another design consideration is the level of threat. Areas with a high likelihood of future development are a priority for protection because they will benefit the most. Remote areas with a low likelihood of development are a lower priority, since protecting these areas will make less of a difference to biodiversity outcomes. That said, remote areas rarely remain free from development indefinitely; development just proceeds at a slower pace.

Finally, the reserve design process should seek to minimize conflicts with other land-use objectives. The rationale here is that a prospective reserve will provide no conservation benefit at all if resource conflicts preclude its establishment.

Generating Design Options

Modern conservation planning initiatives generally use planning software to generate reserve design options for consideration. Several software packages are available, including Marxan (Game and Grantham 2008), C-Plan (Pressey et al. 2005), and Zonation (Moilanen et al. 2014), among others.

To apply planning software, the planning area must be subdivided into discrete cells referred to as "planning units." These serve as the units of selection. The composition of each cell is tabulated to establish its contribution to the representation of each biodiversity element. Other attributes, such as intactness, level of threat, and amount of conflict with resource development are usually tabulated as well, permitting their inclusion in the selection process.

Box 8.2. Planning Unit Options

Several options are available for defining planning units. The most common choice is a grid of hexagons or squares, with a cell size of 1,000 ha or less. The benefit of using a fine-scale grid is that it provides maximum flexibility for assembling efficient reserve designs.

A less common alternative is to use watersheds or other ecological units as planning units. The benefit is that each planning unit is ecologically meaningful. The limitation is that the planning units tend to be large, reducing flexibility. This approach is best suited for planning at very broad scales (e.g., national scale).

A final option is to use administrative units, such as townships, as planning units. This approach may be used if key datasets have been organized by administrative unit.

When planning software is run—using Marxan as an example—it assembles reserves by adding one planning unit at a time until all biodiversity elements have been represented to the desired level (Fig. 8.7A). The selection of planning units is guided by an optimization algorithm. The program seeks to achieve representation targets while simultaneously meeting other design objectives. For example, the program can be set to give priority to cells with a high level of intactness or low resource conflict, while still achieving representation targets. The average size of reserves can be increased by favouring planning units that are adjacent to previously selected cells. Cells that contain biodiversity elements not found elsewhere are said to have high **irreplaceability**, and they tend to be consistently selected. Cells within existing reserves are incorporated automatically, and their contribution to representation is fully accounted for.

The planning units selected in a given model run constitute a potential reserve design. Usually, many alternative designs are possible because the representation targets can be achieved using different combinations of cells. The full spectrum of alternatives can be explored by running the planning program repeatedly. By pooling the results over hundreds of runs it is possible to identify planning units that represent a high priority for protection (Fig. 8.7B).

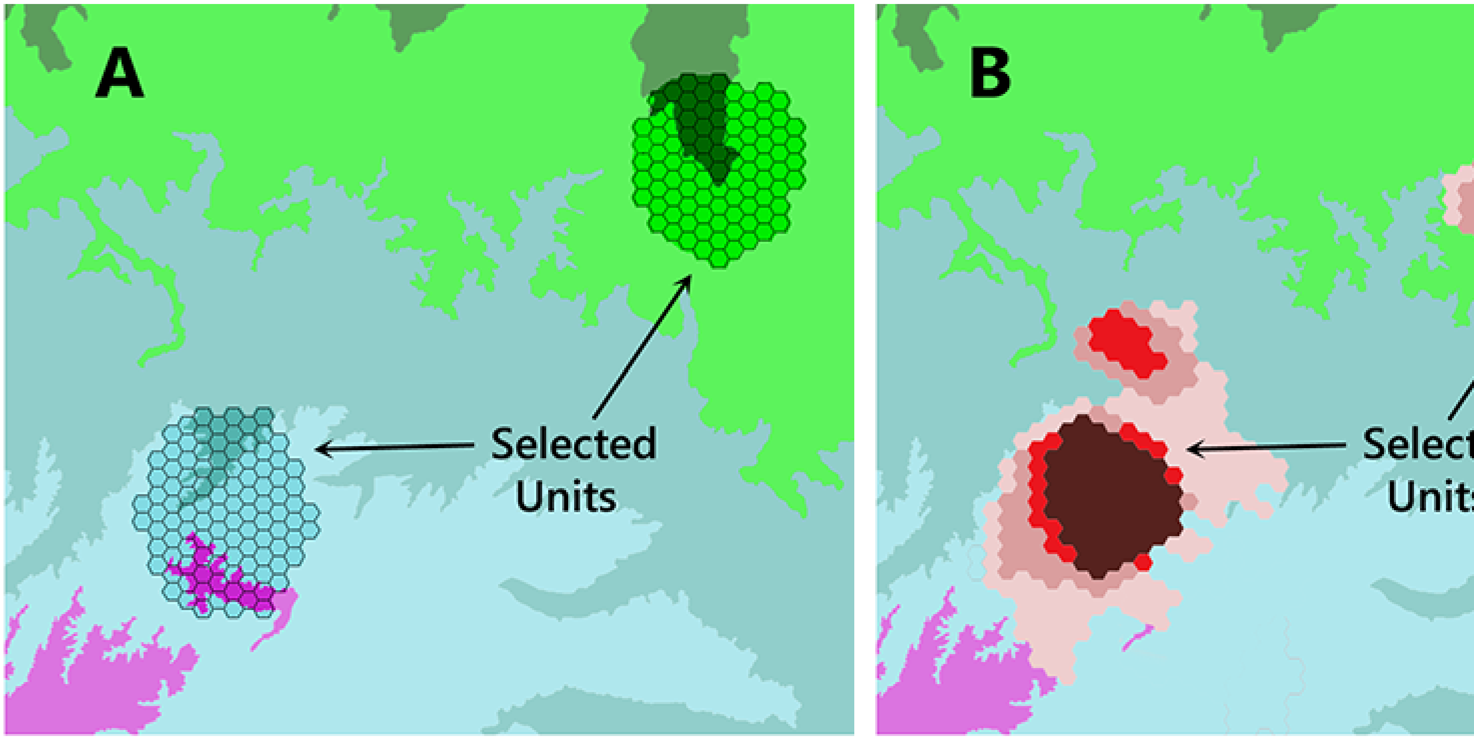


Fig. 8.7. An example of the type of output provided by Marxan, adapted from Case Study 6. Map A illustrates selected in a single model run, overlaid on a map of major ecosystem types. Map B illustrates the summarize model runs, each of which varies slightly. Darker shades of red indicate a higher frequency of selection. Each planning unit is 500 ha.

Connectivity among reserves is difficult to incorporate in reserve planning software and so it is usually addressed manually. What makes connectivity challenging is that priorities for connectivity do not become apparent until after the location of core reserves has been established. There are also decisions about the length and width of connecting zones, and trade-offs with other design features, that cannot be distilled down to a set of simple selection rules.

The main utility of reserve planning software is to support learning about a given system. By running a series of planning scenarios with different combinations of inputs, insights can be gained into design options and the trade-offs that exist among design attributes. A useful starting point is a base scenario that includes only coarse-filter elements. Other elements and design options can then be added systematically, so that their specific influence on the design can be understood. Case Study 6 provides a working example.

Selecting the Optimal Design

Selecting the best reserve design means determining the optimal balance among competing design attributes. There are several trade-offs that need to be considered.

A commonly encountered trade-off involves the representation of focal species versus coarse-filter elements. The more we skew a design in favour of a small set of focal species, the less effective the design will be for protecting overall biodiversity. Underlying this trade-off is a deep question about the purpose of conservation, which we touched on in our discussion of conservation triage in Chapter 6. Do we value all species equally, in which case the coarse filter would be preferred? Or do some species matter more than others, in which case preference would be given to focal species? This is not a question that can be answered through science. It is a matter of social choice and requires input from stakeholders and the public.

There is also a trade-off between representation and functional effectiveness (Rothley 2006). The longer the list of conservation elements to be represented, the less design flexibility there is for achieving functional objectives. Moreover, when many features must be represented, reserve designs become unavoidably fragmented. The maintenance of ecological processes favours reserve systems comprised of large, well-connected sites (Leroux et al. 2007).

It follows that restraint should be exercised when defining ecosystem components under the coarse-filter approach. Just because detailed spatial datasets have become widely available does not mean that representation targets should be set for ever-finer ecosystem elements. The conservation benefit of fine-scale representation is unlikely to offset the resulting loss of design flexibility and reserve fragmentation. Designs that feature many small reserves also carry a high administrative burden.

Another trade-off involves past and future human disturbances. Undisturbed areas are generally preferred because they provide the best starting point for maintaining ecological function. But habitats facing a high risk of future degradation are most in need of protection (Noss 2000). In practice, these two objectives tend to conflict. Intact landscapes are most often found in areas of low economic potential, with an implied low level of threat (Pearson 2010). Conversely, areas with high economic potential, representing a high level of threat, are often highly disturbed.

Finally, there are trade-offs between conservation objectives and resource development objectives (Naidoo et al. 2006). In the past, these trade-offs were often ignored by conservation planners. But experience has shown that conservation designs that ignore socio-economic trade-offs are unlikely to be implemented (Knight et al. 2008). And designs that are not implemented protect nothing. There is, of course, no guarantee that workable design solutions can be found. Nevertheless, it is worth looking because even partial solutions can be helpful.

The exact nature of the trade-offs will depend on local circumstances and there are no generic rules to specify what should be done. Nor does reserve planning software provide the answers—its role is to provide insight into the trade-offs and options that exist (Sarkar et al. 2006). It falls to the planning team to assess the options and to identify a design that best achieves the conservation priorities for the planning area.

Box 8.3. Summary of Reserve Design Trade-Offs

Reserve design is an exercise in optimization across multiple dimensions. The competing objectives generally do not present as binary choices, but as matters of degree (i.e., more of one implies less of the other). In most cases, conflicts are greatest when the amount of land available for protection is highly constrained.

1. **Biodiversity vs. development.** Conflicts between habitat protection and industrial development are almost always present. The level of conflict will vary across the landscape because conservation value and resource potential are usually both unevenly distributed.

2. **Species vs. species.** Species do not all share the same habitat requirements. If a reserve design is biased in favour of a few high-profile species, the habitat of other species may receive less protection.

3. **Representation vs. function.** Increasing the number of elements to be represented tends to produce increasingly fragmented reserve designs. Conversely, the maintenance of ecological integrity and persistence of biotic elements favour designs with large, well-connected reserves.

4. **Intactness vs. vulnerability.** Intact areas are preferred candidates for protection because they present the best starting point for maintaining ecological function. However, the need for protection may be highest in disturbed areas, where vulnerability to further degradation is usually greatest.

5. **Potential for success vs. need.** The paradox of protection is that protected areas are easiest to establish in areas where they are least needed. Limiting protection to low-conflict areas is likely to leave important conservation values unrepresented.

6. **Today vs. tomorrow.** Because the climate is changing and species distributions are no longer static, protection objectives for current and future periods may be in conflict (see Chapter 9).

The Social Dimension of Reserve Design

The preceding description of systematic conservation planning may give the impression that reserve selection is largely a technical exercise. In fact, the final determination of what will be protected invariably entails political negotiation. Moreover, the process is a long, drawn-out affair that proceeds in a piecemeal fashion. Consider that the target of protecting 12% of Canada's landmass was set in the 1980s and we only reached that goal in 2020.

The negotiation component of protected area planning is similar to the policy development process we discussed in Chapter 3. There are long periods without apparent change interspersed with short periods of intense activity leading to the designation of one or more new reserves. The tipping point to active planning occurs when sufficient public, local, and political support have been achieved.

There are many pathways that lead to active planning. Many initiatives involve campaigns by conservation groups to protect sites with high conservation value. These groups often spend years (sometimes decades) building public support, developing local allies, and lobbying the government to take action. The Great Bear Rainforest in BC is a recent example. Other initiatives are driven by local communities, most often in the context of Indigenous land-use planning. The Dehcho and Peel Watershed land-use plans are examples (discussed below). Many protected area initiatives are driven by government policy. The federal government's initiative to protect 30% of Canada's marine environment is an example. With policy-driven initiatives, gaining the support of local communities is paramount, and takes considerable time. Finally, provincial and territorial governments often advance protected areas in the context of regional planning initiatives.

In many cases, systematic conservation planning is undertaken by a government parks agency or a conservation organization well in advance of active negotiations. By grappling with trade-offs among reserve design attributes and identifying areas of greatest benefit to biodiversity these efforts establish conservation priorities. Advance planning may also identify opportunities for minimizing conflicts with other land-use objectives while still achieving conservation targets. The resulting map of conservation priorities often serves in an agenda-setting role, much the same way that area targets do.

In principle, the systematic conservation planning approach can be extended to the negotiation phase of reserve planning. But this is challenging to do in practice. The conservation experts capable of handling the technical aspects of conservation planning generally do not have the expertise needed for negotiating with stakeholders. Nor do they have the authority for making land-use decisions. Therefore, it is common for these roles to be divided between a decision-making team and a technical team. This approach can be effective if the teams are well integrated and the overall process is well structured. Unfortunately, many negotiations devolve to political power struggles, and systematic conservation planning becomes sidelined.

Recognizing the political realities of land-use planning, conservation practitioners involved in protected area negotiations should make their recommendations as meaningful and accessible as possible. This includes helping decision makers and stakeholders understand why particular areas were identified as priorities, and what specific conservation benefits they provide. Options for handling key points of conflict should also be described. This requires a deep understanding of the factors influencing the design process, which is acquired through the systematic exploration of inputs.

In summary, protected areas are established through a long-term process that proceeds in fits and starts. Systematic conservation planning provides critical input to the process, mainly by mapping conservation priorities. But there should be no expectation that conservation priorities will automatically be established as protected areas. A broad base of support must first be established, and this may take considerable time and effort. Final decision making concerning site selection is a political process. It is important to have conservation practitioners involved in the process to ensure that biodiversity outcomes are properly considered. Unfortunately, this does not always happen.

Box 8.4. Common Reserve Planning Pitfalls

1. **Reserve design biases.** For optimal conservation of all species, planners must guard against inadvertently biasing the reserve system in favour of certain design attributes. Common biases include emphasizing representation over function and emphasizing high-profile species over broad biodiversity.

2. **Overreliance on computer software.** Planning software is a tool for identifying design options and exploring trade-offs among objectives. The ecological knowledge needed for developing an effective reserve design lies with the planning team, not the software.

3. **Failure to incorporate climate change.** Climate change has important reserve design implications, especially with respect to connectivity, fine-filter conservation, and climate refugia. This dimension of reserve design has yet to be widely implemented (see Chapter 9).

4. **Failure to consider competing values.** The best reserve design is not the one that perfectly meets the needs of biodiversity. Such designs are useful as reference points but are rarely implemented because they tend to be politically infeasible. Instead, the best design is the one that delivers the greatest conservation benefit given existing social constraints. This requires an extension of the optimization process to include competing socio-economic values.

5. **Becoming bogged down in detail.** With the increased availability of spatial datasets and ample computing power, it can be tempting to set representation targets for an ever-increasing list of attributes. This does not come without a cost. Excessive detail can complicate and slow the design process, reduce design flexibility, and produce fragmented designs with low functional integrity. It also becomes increasingly difficult to systematically explore design options and interpret the results.

6. **Ineffective communication.** Protected area planning is ultimately a social process, so design recommendations must be conveyed in socially meaningful terms. A glossy map and technical description of the methodology are not enough. Decision makers and stakeholders need to understand the objectives underpinning the proposed reserve network and why the recommended design is optimal. Furthermore, because negotiations are typically site-based, the specific contributions of individual priority areas should be provided.

Regional Variations

The Agricultural South

Protected area planning varies among regions because of differences in resource potential and social context. The Agricultural South (Fig. 5.5) consists mostly of privately owned land that has been converted to agricultural use. Remaining patches of native habitat are generally small and widely scattered, except within rangelands in the driest parts of Alberta and Saskatchewan. Approximately 4% of this region has been protected, with a strong bias to dry, unproductive rangelands (ECCC 2022).

Governments are reluctant to infringe on private property rights, so most protected area initiatives in the Agricultural South involve voluntary land purchases, land donations, and **conservation easements**. In a conservation easement, a landowner continues to own the land but agrees to permanent usage restrictions that are designed to maintain the ecological integrity of the site in perpetuity (Good and Michalsky 2008). The federal government encourages donations and easements through tax credits, administered under its Ecological Gifts Program. Protection can also be achieved through **conservation offsets** (see Box 8.5), though the necessary frameworks are only now being developed in Canada.

Box 8.5. Conservation Offsets

A conservation offset program is a method for achieving no net loss of habitat in the face of industrial development. The idea is to have resource companies offset unavoidable habitat damage in areas under development with habitat restoration in other areas. For example, an oil and gas company that damages wetlands in the course of operations might seek agricultural landowners willing to restore wetlands on their farms in exchange for monetary compensation. This approach works best in agricultural settings, where opportunities for restoration are readily identified. In principle, the approach could also be applied to public lands by channelling funding from resource companies to government-led restoration programs. However, the matching aspect of the offset concept is difficult to apply in this case.

A limitation of the offset concept is that true habitat equivalency is difficult to achieve. In most cases, restoration efforts can only partially make up for the loss of natural habitat. Moreover, offsets are only intended to last as long as the period of disturbance, so there is no guarantee that the restoration will be maintained. Efforts to avoid and directly mitigate disturbances are therefore preferable to offsets. Conservation offset programs are also challenging to administer. Application frameworks in Canada are rudimentary at present (Noga and Adamowicz 2014).

In recent decades, most protected area programs in the Agricultural South have been led by conservation groups, supported by government, corporate, and public funding. These protected area initiatives are characterized by direct negotiations between conservation groups and individual landowners about specific parcels of land.

The Natural Areas Conservation Program, funded by the federal government and implemented by the Nature Conservancy of Canada and other partners, has been the largest protected area initiative on private lands in Canada in recent years. It illustrates the pace of protection currently possible. Over a ten-year period (2007–2017), the federal government allocated $300 million to the program. Other donors provided matching funds and land donations totalling $580 million. This resulted in the protection of 4,300 km^2 of habitat across the country (NCC 2017). To put this in perspective, 4,300 km^2 is considerably less than the area of greater Toronto.

The basic aim of protected area planning in the Agricultural South is still centred on identifying optimal sites for protection. However, comprehensive ecosystem representation is difficult to achieve because of the agricultural conversion and urban development that has occurred, and because of constraints on land acquisition. This makes formal reserve design impractical. Instead, protection efforts tend to be opportunistic, guided by simple landscape scoring systems.

In a scoring system, the planning area is divided into planning units and the attributes of each cell are tabulated (Riley et al. 2007). Common attributes include the degree of intactness, habitat value for species at risk, connectivity, vulnerability to degradation, and the cost of acquisition. Instead of using planning software to generate potential reserve designs, the attributes of each planning unit are combined into a composite score reflecting its value to conservation. The results are displayed as a map of conservation priorities which is used to guide land acquisitions.

The benefit of the scoring approach is that the maps are relatively easy to produce. The main shortcoming of this approach is it does not ensure that individual features are represented in alternative design configurations. Nor is it possible to explore trade-offs among designs. Because of these deficiencies, the scoring approach should only be used where systematic conservation planning is not feasible.

An added dimension to conservation planning in the Agricultural South is that protection is added incrementally, as funding becomes available and as willing sellers and donors are identified. This makes planning an ongoing process (Meir et al. 2004). Once a parcel of land has been protected it may serve as the nucleus for an expanding reserve, changing the conservation importance of surrounding parcels. Opportunities may also arise to connect parcels, using restoration if necessary, once sites in reasonable proximity to each other have been protected.

The Industrial Forest

The Industrial Forest (Fig. 5.5) consists mostly of public lands that have been allocated to resource companies engaged in forestry, oil and gas development, and mining. The ecosystems here are no longer pristine, and access development is extensive; however, most ecosystem components and processes remain in place. Approximately 12% of the Industrial Forest has been protected to date (ECCC 2022), with much of this protection biased toward lands of low resource value.

Provincial governments have primary jurisdiction over most lands in the Industrial Forest, so they oversee most protected area planning in this region. The federal government is engaged to a lesser extent through its ongoing efforts to establish national parks. The timing of active planning is irregular, and efforts are not coordinated among provinces. As previously noted, the tipping point to action is highly dependent on public interest and support.

The defining feature of protected area planning in the Industrial Forest is the dynamic that exists between conservation groups, promoting protection, and the resource sector, promoting development. Indigenous communities are also involved, though their role is not as prominent as it is in the Far North. In the Industrial Forest, Indigenous communities are likely to be treated as stakeholders, rather than co-management partners, though there are exceptions. Other common stakeholders include local communities, municipal governments, and various user groups (e.g., trappers, outfitters, tourism operators, and hunting and angling groups).

Conservation planning methods in the Industrial Forest are highly varied. Systematic conservation planning is widely applied, though the process tends to be interpreted differently in each application. As with the ecosystem management framework we discussed in the previous chapter, core elements are often omitted in practice. Furthermore, not all protected area initiatives seek to achieve comprehensive representation. Some initiatives are designed to protect specific sites of high conservation value.

The Far North

The Far North (Fig. 5.5) is comprised of public lands that, for the most part, remain ecologically intact. Industrial activity is limited to widely scattered mining installations and some localized oil and gas development. To date, approximately 14% of this region has been protected (ECCC 2022).

In the past, the planning and establishment of parks in the Far North was led by the federal government. Many of the large national parks and wildlife refuges that exist in this region are the result of these efforts. Over the years, the responsibility for land management has shifted to the territorial governments and Indigenous communities, though the federal government remains involved.

An important driver of protected area establishment in recent years has been land-use planning related to land claim settlements. An example is the Peel Watershed land-use plan, which covers approximately 68,000 km^2 in the northern Yukon (PWPC 2011). The plan was completed in 2011 and it placed a priority on “protecting and conserving ecological and heritage resources and maintaining wilderness character” (PWPC 2011, p. vii). Eighty percent of the planning area was designated as a conservation area and 20% was designated for integrated resource management.

The Peel land-use plan was prepared by an independent planning commission, as per the terms of a land claim agreement signed in 1993 (SCC 2017b). The Yukon government’s reaction to the commission’s plan was mostly negative. It argued that the plan was too restrictive and it unilaterally created a new plan which shifted the balance of protection to 71% industrial and 29% protected. In response, a court case was initiated by a coalition of local First Nations and conservation groups. The case was eventually referred to the Supreme Court of Canada, which ruled in 2017 that the Yukon government could not disregard a planning process it had agreed to under the land claim agreement (SCC 2017b). The commission’s original land-use plan is now going forward to final consultations.

The lesson from the Peel Watershed is that, while much of the Far North remains relatively pristine, protection is still far from easy. As in other parts of Canada, northern governments actively pursue industrial development to provide jobs and tax revenue. This provides the resource industry with considerable political influence, since mining and oil production are often the only source of private sector employment and investment.

Further industrial development in the Far North appears inevitable over the long term, though the pace will be constrained by high transportation costs and the absence of infrastructure. For example, through its *Plan Nord*, Quebec plans to invest almost $2 billion on infrastructure development and other steps to attract new mining projects to the far north of the province (GOQ 2015).

Despite the challenges that exist, the Far North still presents the best opportunity in Canada for establishing additional large terrestrial protected areas. Ontario and Quebec are both acting proactively and have set a 50% protection target in their current northern planning programs (OMNRF 2015; GOQ 2015). Government-led protected area initiatives in the territories are much less ambitious at present. But in these regions, the potential exists for large-scale Indigenous-led initiatives, as exemplified by the Peel Watershed land-use plan.

Marine Environments

Efforts to develop a network of marine protected areas began as a response to the *Canadian Biodiversity Strategy* in 1995 (EC 1995). A national framework was released in 2011 to provide planning guidance (GOC 2011b). More recently, the federal government has committed to protecting 30% of Canada's coastal and marine areas, in accordance with the target established under the UN Convention on Biological Diversity (GOC 2022b).

The federal government has been leading the planning efforts, reflecting its jurisdiction over marine waters under the *Oceans Act* and other legislation. Provincial and territorial governments have been partners in the process and there has been considerable outreach to Indigenous communities and other stakeholders.

Thirteen marine bioregions have been defined and these provide the basis for planning (GOC 2011b). Because of their size, the Great Lakes are included as one of these bioregions, even though they contain fresh water. The reserve network is intended to provide representation of each of these bioregions, emphasizing sites with high ecological significance.

The proportion of marine territory currently protected is 9.1% (524,000 km^2; ECCC 2022). Most of the sites were designated in the last five years. An additional 4.8% of marine territory has been designated as marine refuges where certain industrial activities and harvesting of biological resources are allowed as long as conservation goals are met (ECCC 2022).

The selection and management of marine protected areas is subject to the same trade-offs we discussed in the context of terrestrial reserves. The primary challenge is finding an appropriate balance between protection and economic development. A national advisory panel, commissioned by Fisheries and Oceans Canada, recommended that industrial activities, such as bottom trawling, oil and gas development, and mining be prohibited within all marine protected areas (Bujold et al. 2018). However, the door to development was left open in marine refuges, where industrial activities are allowed if effectively mitigated. Currently, marine refuges account for more than a third of the area that the federal government is counting toward its 30% protection target, and more are planned.

In conclusion, the marine protected area initiative marks a new phase in marine conservation. Much will depend on the total area afforded full protection (vs. refuges) and the locations selected for protection. The marine refuge concept holds many parallels to the sustainable forest management paradigm. There is likely to be improved conservation compared to conventional management, but refuges are not an alternative to true protected areas.

Other Variations

There are also forms of protected area planning that are not based on the systematic representation of biodiversity. These approaches have limited application, but conservation practitioners should be aware of them. A prominent example is the use of biodiversity **hotspots** to guide site selection. Hotspots are areas of particularly high species richness or endemism (Myers et al. 2000). This approach is most commonly employed at the global scale by international conservation organizations and their donors seeking to direct their limited resources to regions that provide the greatest conservation benefit (a form of conservation triage).

Species richness in Canada is relatively low compared to many other parts of the world, so we have no areas rated as globally significant hotspots (Fig. 8.8). However, there are many sites across the country that have long been recognized as having special conservation significance at the regional scale. In most cases, these sites stand out because of their unique characteristics, not because they have higher levels of species richness than other areas. They are best treated as fine-filter elements under a systematic conservation planning framework.

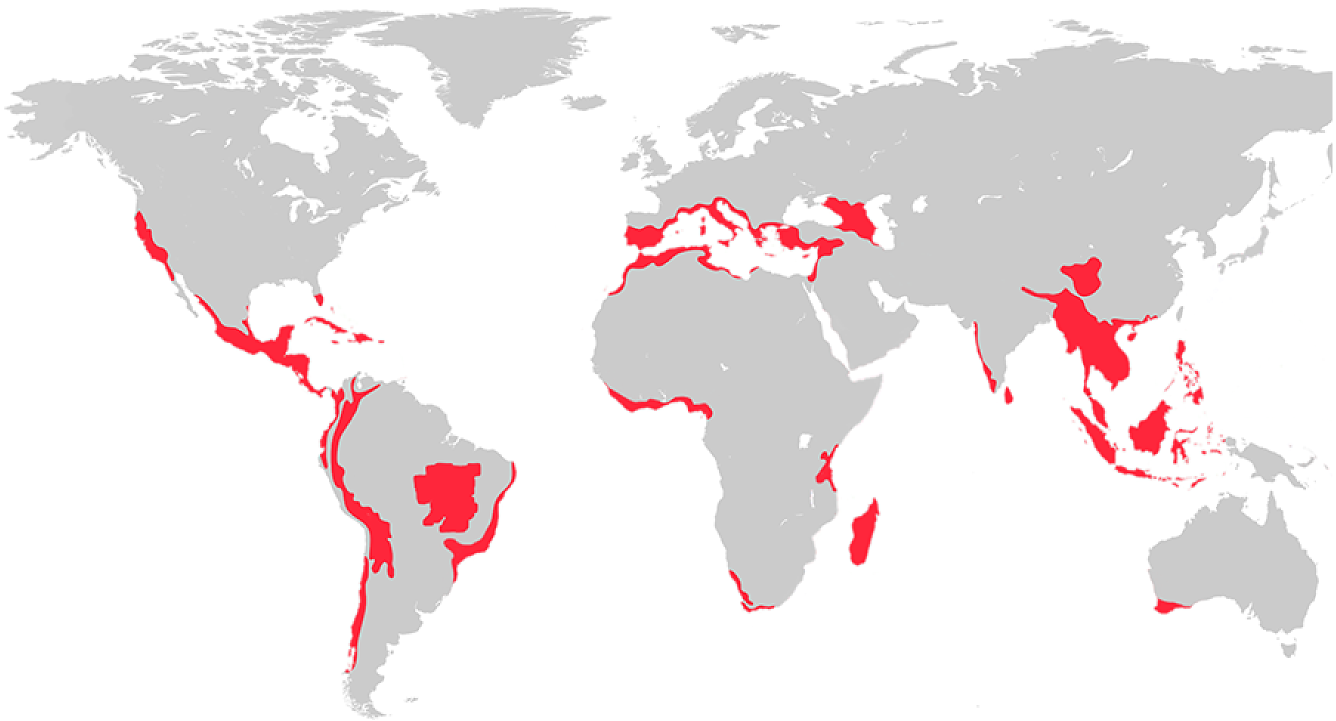


Fig. 8.8. The distribution of global biodiversity hotspots, adapted from Myers et al. 2000. Hotspots are often used by international organizations for allocating conservation effort. At the national scale, biodiversity conservation is best achieved using a systematic conservation planning approach, ensuring that each country's full complement of native species receives protection. Map credit: Ninjatacoshell.

Another approach to protected area planning involves "**floating reserves**" (Rayfield et al. 2008). This approach is usually encountered in the context of forestry operations, where it is promoted as a way of achieving conservation objectives while minimizing economic impacts (Gurmendi 2004). The idea is to protect a constant proportion of the land base using reserves that shift location over time. This approach provides industry with eventual access to the full land base.

The floating reserve concept has received little support as a replacement for permanent protected areas and systematic conservation planning (Rayfield et al. 2008). Over time, it requires reserves with a high level of ecological integrity to be replaced with sites that have been subjected to industrial development. No consideration is given to the decline in condition that inevitably accompanies resource extraction and associated infrastructure development. The claim of protection is therefore illusory.

Where floating reserves do have merit is in the protection of old-growth forest (and dependent species). In regions subject to large wildfires, the location of old-growth stands changes over time, so static reserves may not provide adequate protection. Additional protection can be achieved by setting explicit targets for old-growth retention in forest management plans and achieving these targets using a system of floating old-growth reserves (Kneeshaw and Gauthier 2003). In such a system, old-growth reserves that are harvested or burned are continually replaced by new reserves elsewhere on the landscape. This requires long-term, spatially-explicit harvest planning, and periodic replanning, to ensure that bottlenecks in the supply of old-growth do not occur over time.

Managing Reserves

Threats

Although industrial development is prohibited within protected areas, most reserves are expected to support tourism and recreation, and many allow hunting, fishing, and trapping (Fig. 8.9). These activities are not necessarily incompatible with conservation objectives; it is the type, intensity, and extent of use that determines the impact (Pickering 2010). For example, there is a significant difference between hiking along designated trails and unrestricted all-terrain vehicle use. The amount of infrastructure needed to support tourism and other activities is also an important factor. Roads, in particular, are a serious threat to biodiversity, given their many deleterious effects (see Chapter 5).

Fig. 8.9. Parks must strike a balance between maintaining ecological integrity and supporting tourism with its attendant need for infrastructure. Credit: M. Quinn.

Internal threats may also include the legacy of anthropogenic disturbances that occurred prior to reserve establishment. This is a common problem with new protected areas in the Agricultural South, where ecosystems have often been converted to agricultural use. In northern areas, the cumulative legacy from earlier mining, oil and gas, and forestry operations is often a concern. Legacy effects can also include changes in species composition and age structure resulting from long-term fire suppression (Baker 1994).

External threats, arising from industrial activities in the adjacent working landscape, are also a concern. Common examples include water pollution flowing in from upstream mines and mills, acid rain, and altered water flow patterns from dams. The working landscape can also act as a sink for wildlife populations, because dispersing animals may experience increased mortality upon leaving the reserve (Wiersma and Simonson 2010). The level of threat rises with the degree of habitat degradation in the working landscape, especially in the immediate vicinity of the reserve. Small reserves face the greatest risk because they have the most exposed edge relative to interior habitat (Fig. 8.6). Finally, invasive species and climate change can also be of concern.

Taking Action

The conservation measures used for countering threats to ecological integrity within reserves are similar to those used on the working landscape. To mitigate the impacts of recreation and other visitor activities, managers can place restrictions on the types and amounts of activities that are permitted and where they can take place. To reduce the legacy effects of past anthropogenic disturbances, managers can implement restoration programs, including species reintroductions. Prescribed burns can be used to recreate natural ecological patterns that have been altered by long-term fire suppression (Baker 1994). In severely compromised sites, managers may need to replace the ecological function of missing keystone species through artificial means. For example, they may have to cull species that lack predators or introduce domestic species to emulate natural grazing.

In addition to the direct mitigation of threats, managers can support the maintenance of ecological integrity through public outreach and education. Helping visitors understand the reason for restrictions on recreation and other activities improves compliance (Marion and Reid 2007). In addition, interpretive programs and positive visitor experiences bolster public appreciation for nature. This translates into increased environmental awareness and support for parks and conservation efforts in general.

External threats are more challenging to deal with because park managers have no authority over activities that occur in the working landscape. Managers can appeal to other levels of government for support, and they can engage directly with external land users in collaborative planning initiatives. Special management zones along park boundaries can be helpful in buffering external impacts. Internal programs can be implemented to mitigate unavoidable threats.

For management efforts to be efficient and effective, they should be supported by monitoring (Hockings 2003). Monitoring is also needed to support the ecological benchmark role of protected areas, allowing them to serve as reference points or “controls” for management activities occurring in the working landscape.

The management of reserves is subject to a variety of constraints. Though managers set the rules on permissible activities within reserves, they do not have a free hand. Managers must work within the basic parameters set by the legislation that supports each class of park. Moreover, tourism operators, user groups, and conservation groups often seek to advance specific agendas and expect to be consulted on proposed management actions.

Another important constraint is the availability of funding and technical capacity. Management interventions and monitoring cannot be undertaken without adequate financial support. In most regions, available budgets are considerably less than what is required for effectively identifying and responding to all threats (Lemieux et al. 2011). Therefore, managers usually must prioritize their efforts.

CHAPTER IX
CLIMATE CHANGE

Climate Change

Climate change is expected to cause widespread changes in the distribution of species and ecosystems across Canada. This presents new threats to biodiversity and it also forces us to reconsider the aims of conservation. What does it mean to *maintain* biodiversity in a world of constant change?

We will begin this chapter with a review of the climatic and ecological changes expected to occur in the coming decades. An important theme here is discerning the difference between change and threat. Next, we will consider the changes to conservation objectives and planning methods needed to accommodate climate change. Finally, we will discuss required adjustments to conservation practices. For the most part, adaptation entails using existing tools in new ways and for new purposes.

Canada’s Changing Climate

Global temperatures have been rising over the past century, especially since 1980 (Fig. 9.1). This increase has been attributed to a greenhouse effect arising from the anthropogenic release of CO_2 and other greenhouse gases (Cook et al. 2016). The current level of CO_2 is well above the maximum level recorded over the preceding 800,000 years and is steadily increasing (Fig. 9.2; Lüthi 2008).

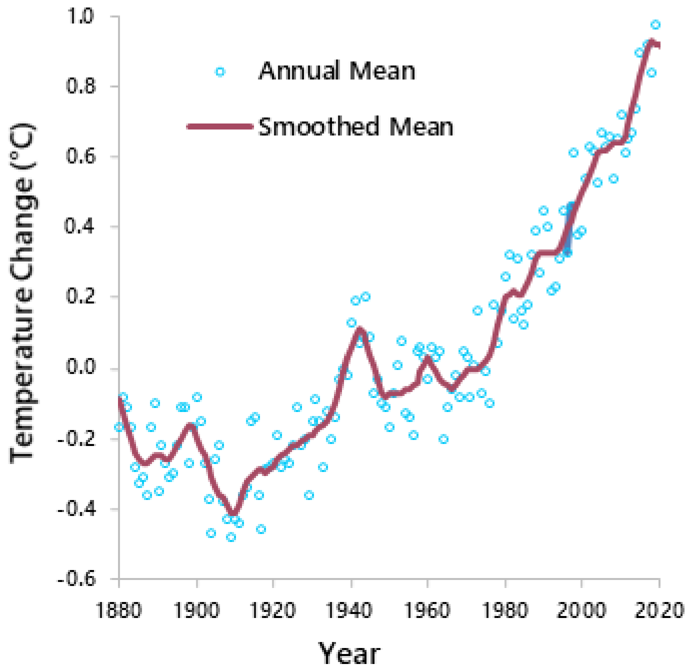


Fig. 9.1. The change in global surface temperature from 1880–2022, relative to the 1951–1980 reference period. Source: NASA 2023.

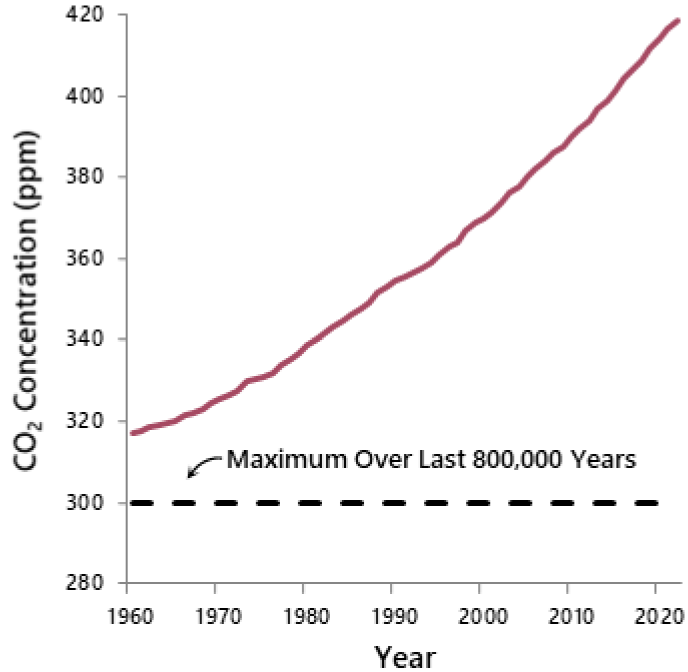


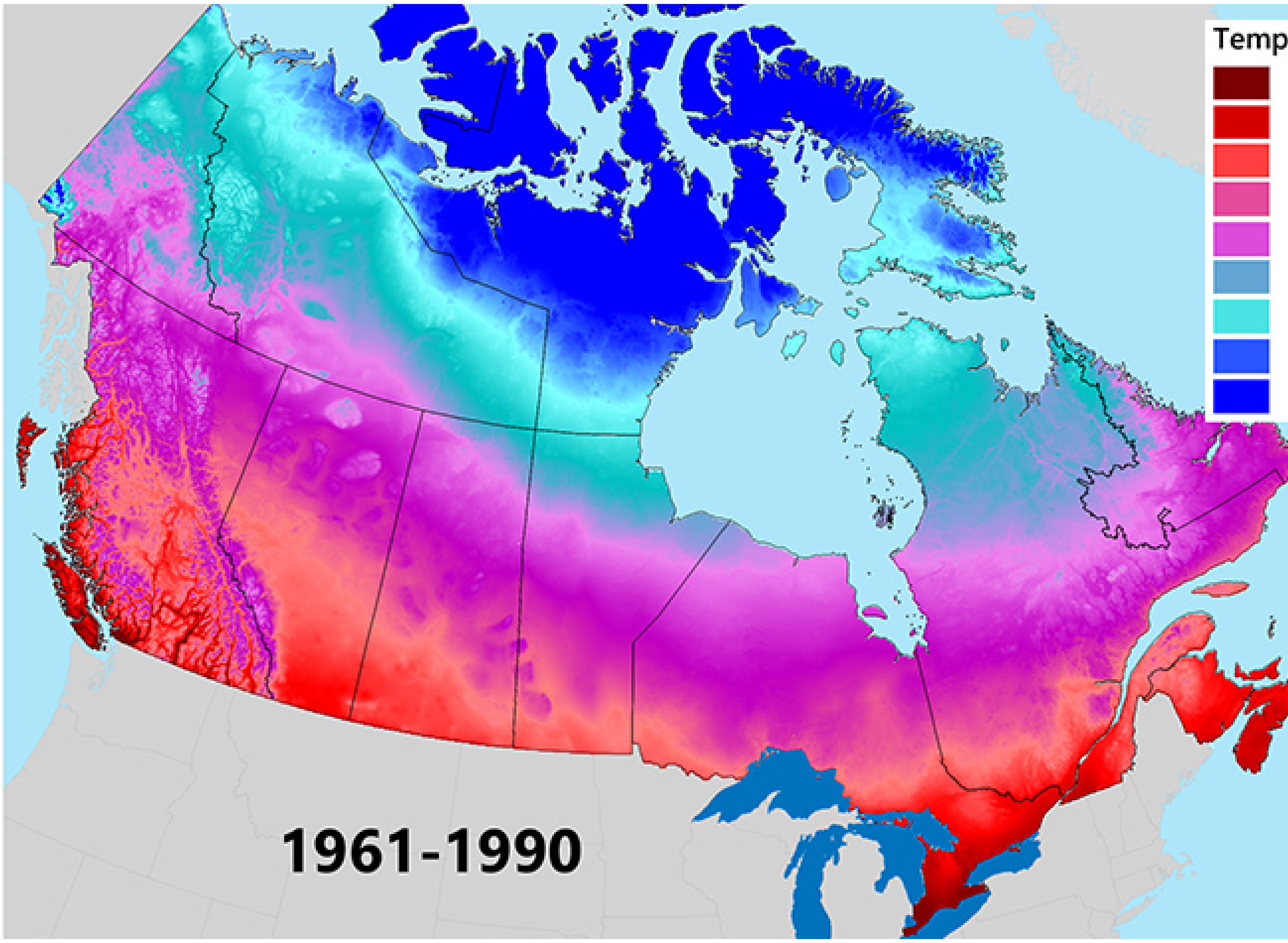
Fig. 9.2. Global CO_2 concentration, 1960–2022. The dashed line indicates the maximum CO_2 level recorded over the preceding 800,000 years. Source: Lüthi et al. 2008; NASA 2023.

The rate of warming in Canada has been almost double the global average because greenhouse gas effects are magnified at high latitudes. Temperatures here are now 1.9°C above mid-twentieth-century norms (ECCC 2023). Regionally, the rise in temperature has been greatest in the northern territories and lowest in Atlantic Canada (ECCC 2023). Seasonally, warming has been greatest in winter and spring.

Trends in annual precipitation have been less discernible than those of annual air temperature. The general trend in Canada has been toward an increase in precipitation, but some regions, such as the southern prairies, have experienced a decline in recent decades, amid much year-to-year variability (Mwale et al. 2009). The Arctic has seen rapid declines in sea ice extent, both in summer and winter.

The amount and pace of future warming will depend in large part on how much more CO_2 and other greenhouse gases are released in coming years. This hinges on human behaviour and political processes at the global scale. A global rise of at least 2°C (relative to the late twentieth century) seems unavoidable, given the amount of CO_2 that has already been released (Clark et al. 2016). Worst-case scenarios suggest that more than 6°C of warming is possible if emissions remain unchecked. To put this into perspective, the rise in global mean temperature between the last glacial maximum, 21,000 years ago, and the beginning of the twentieth century was approximately 5°C (Clark et al. 2016).

Figure 9.3 illustrates how temperature patterns are expected to change across Canada under an intermediate level of emissions. Under this scenario, Canada's mean annual temperature is predicted to rise by more than 4°C by the end of this century. Warming will be greatest in winter, and in this season, the largest increases in temperature are projected for northern Canada (Warren and Lemmen 2014). The melting of permafrost, which is already underway, will accelerate. In summer, the largest increases in temperature are projected for southern Canada and the central interior. Warmer temperatures will lead to a longer snow-free period in all areas, and a nearly ice-free summer is considered a strong possibility for the Arctic Ocean by 2050.


Temp
1961-1990

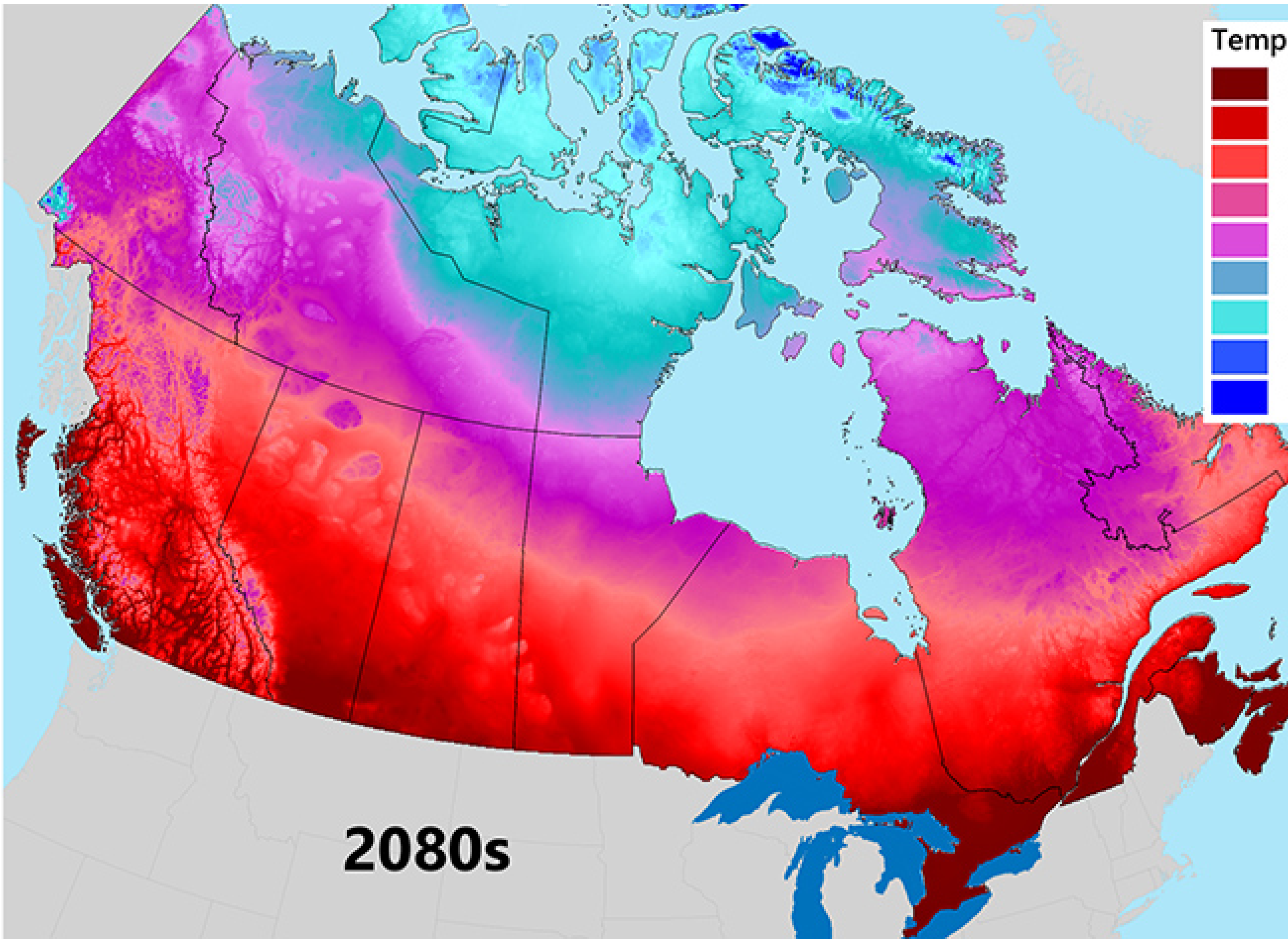


Fig. 9.3. The mean annual temperature in Canada. Top: 1961–1990 reference period. Bottom: projected temp under the second-lowest emission scenario (RCP 4.5; ensemble mean). Source: ClimateNA interpolation, avail https://adaptwest.databasin.org/pages/adaptwest-climatena.

Warmer temperatures increase the rate of evaporation from water bodies and soil and increase transpiration from plants. With more moisture in the air, precipitation will increase, though it will not be distributed evenly. Some areas, especially the southern prairies, are likely to experience a decline in precipitation in summer and fall rather than an increase (Warren and Lemmen 2014; Wang et al. 2016a). In these areas, the combination of lower summer rainfall and increased evapotranspiration from higher temperatures is expected to cause soil moisture levels to decrease during the growing season. In other areas, increased evapotranspiration and increased precipitation are likely to balance each other out, and soil moisture should remain generally stable or even increase (Price et al. 2013).

Warmer temperatures also imply an increase in the overall energy content of the atmosphere. As a result, global and regional weather patterns will become increasingly chaotic. This implies a greater frequency of extreme climatic events, including droughts, floods, and heat waves, affecting all parts of the country (Warren and Lemmen 2014).

Episodes of extremely hot and dry weather are particularly important because these conditions increase the risk of large forest fires (Flannigan et al. 2016). The lengthening of the snow-free period and constraints on fire-fighting capacity will also contribute to increased rates of burning (Wotton et al. 2017). Fire models project that the western boreal forest will be most affected. The average area burned in this region could triple by the end of the century (Balshi et al. 2009). Eastern Canada will also experience increased fire risk, though not as extreme as in the West (Wang et al. 2015).

Warming, particularly in winter, is also expected to increase the frequency and severity of insect outbreaks (Volney and Fleming 2000). The recent spread of the mountain pine beetle east of the Rocky Mountains, where cold winters had previously kept them in check, indicates that this is already becoming a reality (Cullingham et al. 2011).

Ecological Responses

Bioclimatic Envelope Models

The most common approach for predicting species responses to climate change is bioclimatic envelope modelling (Araujo and Peterson 2012; Gray and Hamann 2013). In this approach, a statistical model is used to characterize the climatic conditions within the current range of a species (i.e., its **climate envelope** or space). Once the relationship between climate and range has been quantified, predictions can be made about where the species is likely to be located in future periods, after climatic conditions have changed. The underlying assumption, supported by the paleoecological record, is that a species will track its preferred climate as it shifts through space and time (Martinez-Meyer et al. 2004; Wiens et al. 2010). The same basic approach can be used to predict spatial shifts of entire biomes (Hamann and Wang 2006; Rehfeldt et al. 2012).

One use of bioclimatic modelling is to estimate the **climate "exposure"** of a species. Exposure is a combination of the amount of expected change in the climate envelope and the **velocity of change**, which is the distance that a climate envelope shifts per unit of time (Loarie et al. 2009; Hamann et al. 2015). Faster rates of change present a greater risk because there are limits to how fast a species can shift its range.

Climate sensitivity is a complementary measure that describes a given species' ability to cope with change. It is a function of species-specific adaptability traits. Anthropogenic disturbances that reduce resilience or create barriers to movement are also a factor. Climate exposure and climate sensitivity are combined to produce an overall **vulnerability assessment**.

Bioclimatic projections are also used to support a variety of planning applications, which we will discuss later in the chapter. These include:

- Identifying areas of relative stability that can serve as climate refugia
- Guiding restoration programs
- Designing protected area systems and connectivity networks
- Facilitating the range shifts of individual species

Bioclimatic envelope models are subject to several limitations that conservation practitioners should be aware of (Pearson and Dawson 2003). Foremost, model projections represent the state of the system after biotic elements have had time to equilibrate with the climate. Insight into transitional stages, which will predominate during this century, must be obtained separately through empirical study.

Another limitation of bioclimatic envelope models is that their reliability declines rapidly when applied below the regional scale. This is because factors other than climate, such as soil type, topography, disturbance history, and biotic interactions become increasingly influential in determining ecological patterns as one moves from the regional to the local scale. Furthermore, global climate models do not provide the resolution needed for fine-scale applications.

Bioclimatic envelope models are also subject to all of the limitations of statistical habitat models that we discussed in Chapter 6. In particular, the application of these models to future periods represents an extrapolation to novel conditions, making reliability difficult to judge (Pearson and Dawson 2003).

The paleoecological record indicates that the relationship between climate and ecological distributions is reasonably robust, though not inviolable (Martinez-Meyer et al. 2004). Major ecosystem types appear to have shifted northward largely intact during the Hypsithermal period (approximately 6,000 years ago), when temperatures in Canada increased by 2–3°C (Dyke 2005; Strong and Hills 2005). This suggests that bioclimatic envelope models should be reliable under similar levels of warming. But existing bioclimatic relationships may well break down under higher levels of warming, as may occur under median- to high-emission scenarios. In short, the warmer it becomes, the more uncertainty there is about how ecological systems will respond.

Vegetation Responses

Bioclimatic envelope model projections indicate that species and ecosystems in Canada will shift significantly northward and upslope under even the lowest CO_2 emission scenarios (Hamann and Wang 2006; Gray and Hamann 2013). The exact amount of ecological change will vary by region and will depend on how much warming actually occurs. Changes will be most apparent along ecotones and in parts of the country subject to both warming and drying.

The Prairie Provinces and the Territories are expected to experience the greatest ecological shifts. A case in point is northern Alberta, which receives just enough moisture to support a closed canopy forest (Hogg and Bernier 2005). Increased evapotranspiration from higher temperatures is expected to cause a net decline in moisture levels, leading to the eventual transition of most of Alberta's boreal forest to parkland and grassland (Fig. 9.4; Stralberg et al. 2018). Changes will be much less extreme in BC and Eastern Canada because precipitation inputs are higher, making it unlikely that forests will become moisture stressed under warmer conditions (Price et al. 2013). In fact, these forests are expected to become more productive and more diverse in the future (D'Orangeville et al. 2016). Grassland regions will experience a change in species composition but are expected to remain as grassland ecosystems.

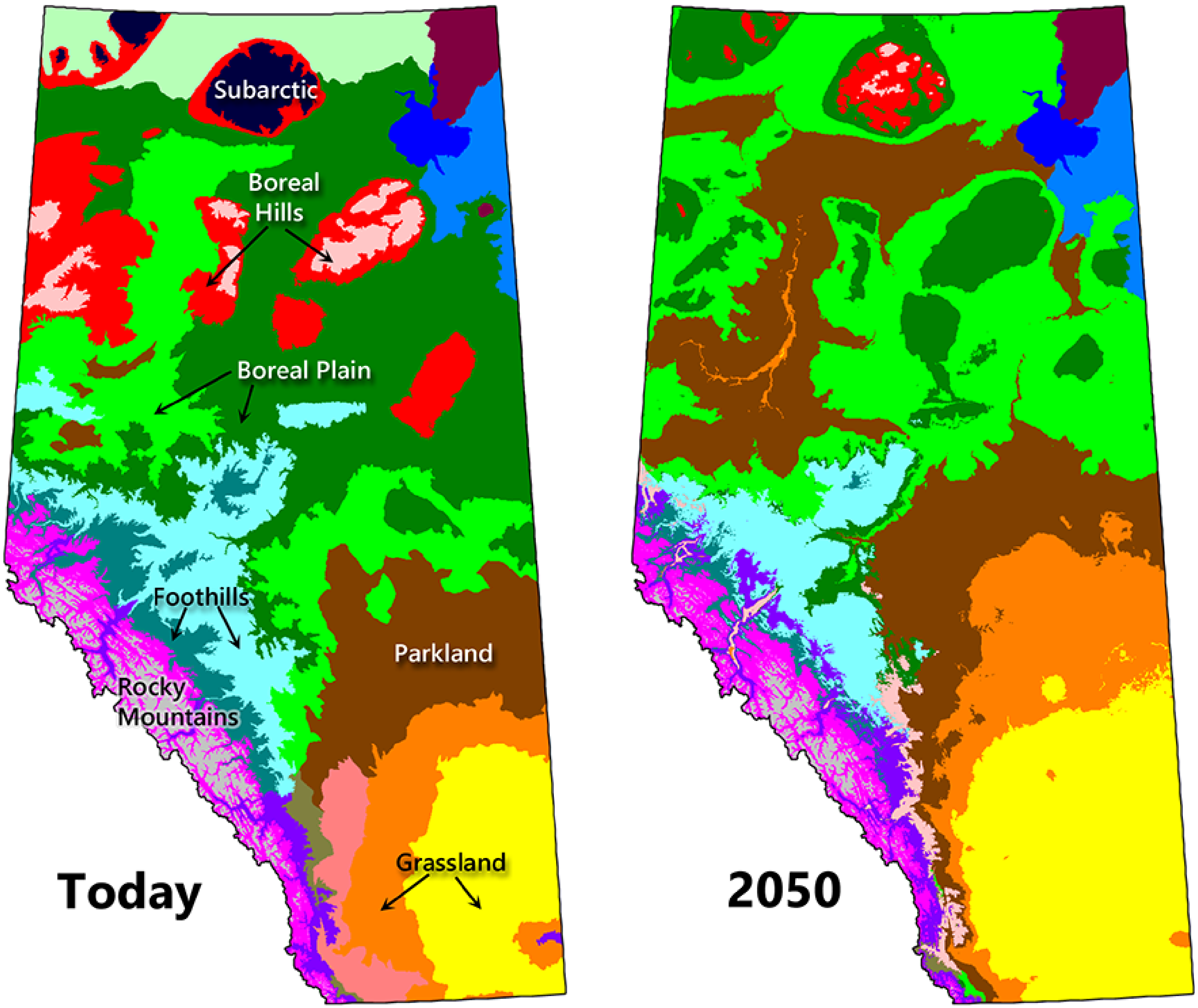


Fig. 9.4. The current distribution of Alberta's major ecosystem types (left) and bioclimatic envelope model projections of these ecosystems for the 2050s (right) under an intermediate climate change scenario (ECHAM5-A2). The projections reflect changes in the climate envelopes; ecosystem transitions will lag substantially behind. Adapted from Schneider and Bayne 2015.

In addition to knowing the long-term trajectory of change, it is important to understand the mechanisms by which the ecological transitions will occur. Computer modelling is not yet an option, though dynamic vegetation models that include climate change are under development (Fisher et al. 2015a). For now, we must draw directly on empirical research and ecological first principles to gain insight into transitional processes.

Plants have a certain amount of resilience to climatic changes, both at the individual level, through structural and functional plasticity, and at the population level, through genetic variability (Hof et al. 2011; Reed et al. 2011). However, these mechanisms will not be sufficient to accommodate the large rise in temperature that is anticipated. Plant species will also need to shift their ranges (Corlett and Westcott 2013; Savage and Vellend 2015).

Climate-induced range shifts will occur through differential reproduction and mortality along range margins. Where climatic conditions are improving for a given species, referred to as the **leading edge**, competitive ability will be enhanced and mortality will be reduced (Koen et al. 2014b). Over time, this will lead to increased abundance of the species along the leading edge, and the colonization of new landscapes ahead of it. Similar processes will occur in reverse along the **trailing edge** of the range, leading to lower abundance and abandonment of range (though not necessarily at the same rate). The net effect is a slow directional shift in range.

It is important to note that climatic effects will not be limited to changes at range margins. Warming will disrupt the competitive balance that exists among species in all areas (HilleRisLambers et al. 2013; Hargreaves et al. 2014). From paleoecological data and contemporary ecological studies (Martinez-Meyer et al. 2004; Wiens et al. 2010), we know that the natural balance among species will seek to re-establish itself over time, as competitive interactions exert their inexorable effect (subject to caveats discussed below). Consequently, an observer in a fixed location will see a gradual change in the relative abundance of species over time as this rebalancing takes place (Kelly and Goulden 2008). At the macro scale, these changes will manifest as a slow directional shift in the distribution of entire plant communities along the prevailing climatic gradient (Fig. 9.5).

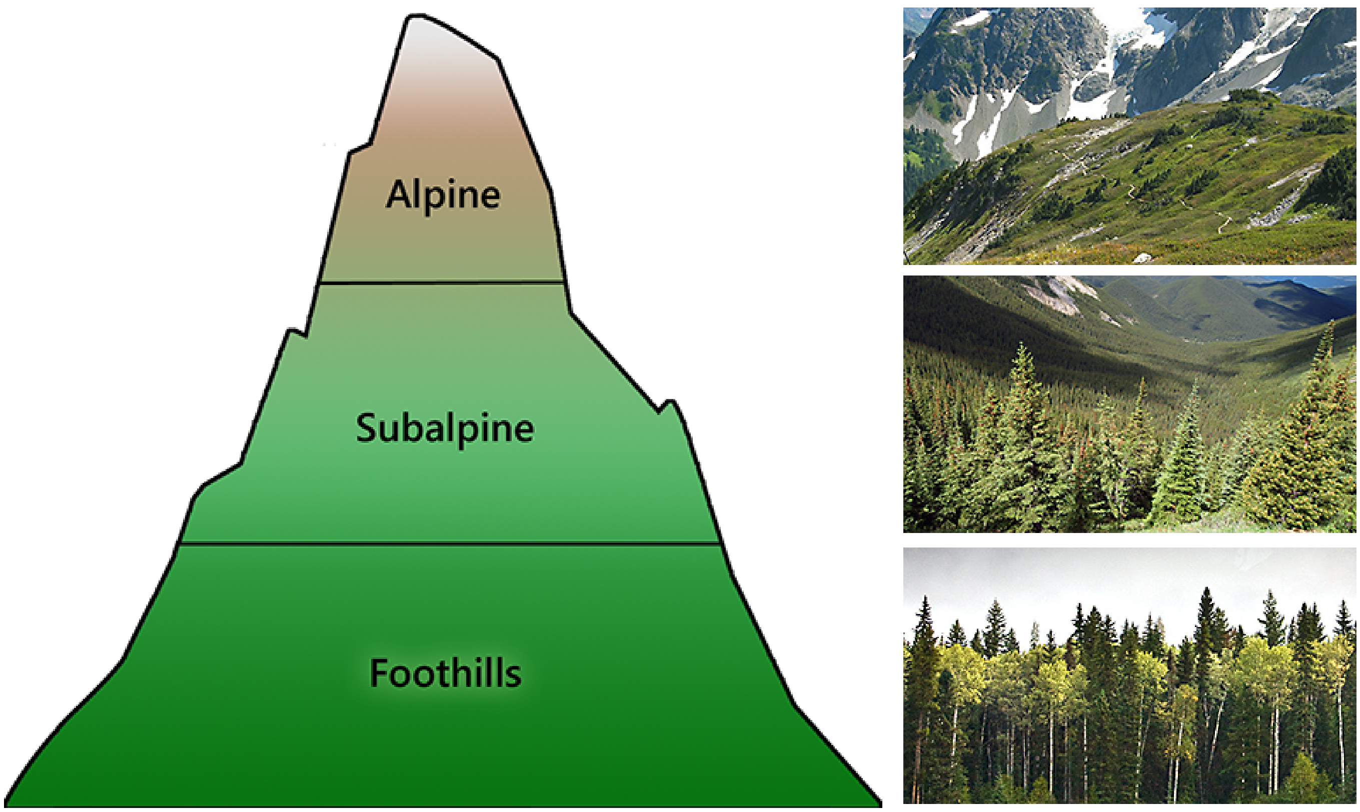


Fig. 9.5. The influence of climatic gradients on ecological distributions are easiest to observe in mountainous areas, where changes occur across relatively short distances. Moving upslope, mixed coniferous and deciduous forests give way to a sequence of coniferous forest types, and eventually, to alpine meadows that are largely devoid of trees. The elevation at which these transitions occur depends on latitude, indicating that it is mainly temperature, not elevation, that is responsible for the observed pattern. Photo credits: Foothills: J. Pang; Subalpine: P. Krömer; Alpine: D. Hershman.

As a general rule, ecological transitions will lag significantly behind changes in the climate. Ecological systems have inertia, much of it due to the "priority of place" (Suttle et al. 2007; Hille- RisLambers et al. 2013). Established plants are not easily displaced by new arrivals, even if the climate has become suboptimal for them (Urban et al. 2012).

Superior competitors will eventually prevail but they may need to wait for a window of opportunity provided by the mortality of existing plants. This may take decades in forest systems.

Aspen provides a useful example. Recent surveys along the eastern slopes of the Rocky Mountains have found aspen seedlings growing as high as 1,500 m, which is 200 m higher than previously recorded (Landhäusser et al. 2010). All of these seedlings were growing within forestry cutblocks, which provided the disturbed conditions needed for aspen establishment. Aspen was unable to establish within adjacent mature forest stands, even though climatic conditions were similar. The implication is that the potential range of aspen has already expanded as a result of the warming that has occurred in recent decades. However, the utilization of this new range is being impeded by the presence of existing vegetation. Most species are likely to experience these sorts of lag effects (Bedford et al. 2012; Gray and Hamann 2013).

The aspen example demonstrates that disturbance rates will be a key factor governing the rate of ecological transitions. Anything that kills or seriously weakens existing vegetation, including fire, severe drought, insect outbreaks, or windthrow, will provide a window of opportunity for competitive rebalancing and range shifts. Conversely, areas that remain undisturbed will transition much more slowly, despite progressive changes in climate. As a result, instead of progressing as a wavefront, ecological transitions will be patchy and widely distributed, reflecting the scattered distribution of natural disturbances (Fig. 9.6). The types of transitions that occur within disturbed sites will depend on how much climatic warming has occurred to that point and on local factors such as the availability of seed sources.

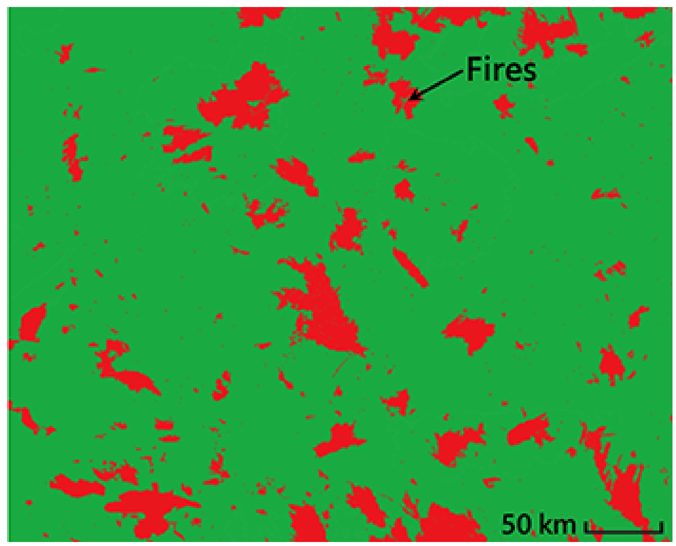


Fig. 9.6. Climate-induced ecological transitions will exhibit a patchy distribution reflecting the occurrence of fire and other major disturbances, as illustrated by this map of fires in northern Alberta over the last 50 years. Moreover, ecological transitions will not occur until after the climate envelope has shifted significantly from its historical norm. Source: Alberta Wildfire Database.

Another factor that limits vegetation responses to climate change is dispersal ability (Urban et al. 2012). The rate that plants can invade new landscapes, given suitable conditions, depends on the distance their seeds can travel and the time it takes for seedlings to mature and produce seed. Given the extraordinarily fast rate of climate warming, few species will be able to disperse fast enough to keep pace with the leading edge of their climatic envelope (Corlett and Westcott 2013). The ones that do are likely to be pioneer species, like fireweed, with adaptations for exploiting new and potentially distant habitats. A corollary is that climate change is expected to facilitate the expansion of invasive species, both native and alien (Walther et al. 2009).

Because species have different rates of dispersal, ecosystems will not shift as intact units. Instead, we can expect novel combinations of species to arise during the transitional period, as some species race ahead and others lag behind (Urban et al. 2012). The faster the rate of warming, the greater these anomalies will be.

Changes in ecosystem composition are also expected to arise from differences in species resilience. For example, feedback processes within peatlands provide resilience to warming and drying (Waddington et al. 2015). As a result, large peatlands will likely persist in the western boreal forest well into the next century, despite increasing moisture deficits (Schneider et al. 2016). In contrast, white spruce is expected to steadily decline as a consequence of regeneration failure under drier conditions. The resulting transitional ecosystem, dominated by peatlands, aspen, and open grassland, will be unlike anything that currently exists in Canada today.

In summary, we can expect that ecosystems will generally follow the trajectories described by bioclimatic envelope models, but the projected equilibrium conditions will not be reached until well into the next century, or beyond. In the meantime, novel transitional ecosystems will predominate. The rate and spatial pattern of vegetation transitions will depend on both the rate of warming and the occurrence of natural disturbances. Ecosystems will become complex admixtures of old and new elements, blurring ecosystem boundaries and increasing habitat diversity in most regions (Berteaux et al. 2010; Savage and Vellend 2015).

Box 9.1. An Old-Growth Bottleneck

Old-growth forest stands are lost to natural disturbances and forest harvesting, and they are replenished through the maturation of younger stands. The rate of loss will increase under climate change because of an increase in the rate of fire and other natural disturbances (Kuuluvainen and Gauthier 2018). These losses will eventually be offset by the expansion of forests into more northerly regions. However, the rate of old-growth production is expected to be much slower than the rate of loss for many decades, resulting in a critical bottleneck. Maintaining a minimum amount of old-growth forest during this transitional period will require a reduction in harvest rate and careful spatial planning.

Animal Responses

As with plants, bioclimatic envelope models suggest that animal distributions will shift northward and upslope in response to warming temperatures, with considerable individual variability. Again, these are equilibrium projections that will not be realized immediately. Transitional processes will predominate well into the next century.

Climate affects animal distributions through a combination of direct physical effects and indirect effects on habitat supply. Habitat generalists will respond to climate change differently than habitat specialists. For generalists, the direct effects of warming are likely to be most important. For example, white-tailed deer are found across many ecosystem types, from forests to grasslands. The northern extent of their range is thought to be determined mainly by winter severity, which affects survival and reproduction (Dawe and Boutin 2016). Warmer winters are likely to lead to range expansion long before vegetative communities have responded. Evidence of such climate-induced range expansion is already accumulating (Veitch 2001; Dawe and Boutin 2016).

Fig. 9.7. The burrowing owl is an endangered species that is contracting toward the core of its range in the US rather than expanding northward in response to climate change. Credit: B. Garrett.

For most other animal species, habitat suitability is likely to be the main factor governing range shifts (Kissling et al. 2010; Nixon et al. 2016). For example, the climate envelope for the burrowing owl—an endangered grassland species at the northern extent of its range in Canada—is projected to move northward fairly rapidly (Fig. 9.7; Fisher and Bayne 2014). However, the owls will not be able to utilize this expanded potential range until entire grassland ecosystems, including appropriate vegetation and prey species, are in place. The upshot is that, despite their high mobility, many animal species will be unable to respond to climate changes any faster than the plant species they ultimately depend on (Nixon et al. 2016).

An additional problem in adapting to climate change is that many species must simultaneously cope with the effects of industrial development and other human activities. The more a species is impacted by human disturbances, the lower its adaptive capacity for responding to climate change.

The concern is greatest for species that are struggling to remain viable under current conditions. Returning to our burrowing owl example, instead of being primed to expand along the northern leading edge of its range, as conditions become suitable, it is currently declining in abundance and contracting southward toward the core of its range in the US (COSEWIC 2006). Moreover, its small remnant populations in Canada may lack the resilience needed to withstand extreme weather events, which are expected to become more common as the climate warms (Oliver et al. 2013; Fisher et al. 2015b).

Human disturbances can also form physical barriers that impede a species' ability to track its preferred climatic conditions. Habitat fragmentation presents the most widespread barrier, and while not completely blocking movement, it slows the pace of adaptation (Barber et al. 2016). For example, the ability of a grassland species to track its preferred climate as it moves northward is likely to be hindered by the presence of intensively managed agricultural land (Nixon et al. 2016). For fish, stream fragmentation may reduce their access to climatically optimal watercourses (Park et al. 2008).

Indirect Effects

Canada's harsh climate has been a barrier for many non-native invasive species, but this barrier is now weakening. New alien species are expected to become established here, and those that are already present are expected to expand their distribution (Walther et al. 2009; Smith et al. 2012).

Another indirect effect of climatic warming involves changes in human activity patterns. Of particular significance to biodiversity is the potential for agricultural expansion into areas that are currently forested. As we saw in Chapter 5, agriculture is far more detrimental to biodiversity than forestry and other forms of industrial activity. Such expansion is already being contemplated. For example, in their analysis of climate change implications for Saskatchewan, Carr et al. (2004) write:

> As areas in southern Saskatchewan and Alberta become too arid for maintenance of herds of animals, it may be possible to open up new agricultural areas north of Prince Albert. These areas are too cold now, but as the length of the growing season increases, perhaps the present boreal plain can be made the new breadbasket of the country. (p. 26)

Although agricultural expansion would provide important societal benefits, including jobs and food production, it nevertheless represents a serious threat to the conservation of native biodiversity. This is especially true if the transition to agriculture involves the privatization of public land, and concomitant loss of government control over land-use practices.

A related threat to biodiversity is that, as the climate warms, resource companies may demand a loosening of operating rules designed to maintain biodiversity (Carr et al. 2004). Companies may argue that such restrictions are futile—like building sandcastles before the tide—given the impending overriding effects of climate change. But such arguments disregard the importance of supporting species as they adapt to change. In the face of climate change, species need more help, not less.

Change Versus Threat

Though widespread changes in ecological distributions seem inevitable, these changes need to be placed in context. Not all changes are threats. Much depends on the frame of reference used.

Species diversity in Canada declines with latitude, and overall it is relatively low compared with many other countries (Willig et al. 2003). A northward movement of ecosystems under climate change would boost biotic productivity and species richness at all latitudes (Jia et al. 2009; Savage and Vellend 2015). Thus, from the perspective of species richness, climate change may be a generally positive factor for Canadian biodiversity (Berteaux et al. 2010), though there are caveats that we will discuss below.

If we use regional ecosystems as our frame of reference, it is apparent that a warmer climate will result in both “winners” and “losers” (Zhang et al. 2015). For example, alpine and high Arctic ecosystems will be compressed because there is nowhere for these ecosystems to go. Conversely, grassland ecosystems in the Prairie provinces, and mixedwood systems in the Niagara and St. Lawrence regions, are expected to expand into the southern boreal region, making them beneficiaries of a warming climate (Schneider et al. 2016). Net changes in other ecosystems are difficult to predict because they will experience both gains and losses along their leading and trailing edges, respectively. Overall, it is a zero-sum game: losses in one ecosystem must be offset by gains in another. What is notable is that the majority of Canada’s species at risk reside in southern ecosystems that are expected to expand.

During the transitional period that will predominate during this century, ecosystem complexity at the regional scale will generally be increased relative to current levels (Savage and Vellend 2015). As previously discussed, the composition and structure of existing ecosystems will initially be augmented by scattered patches of new elements. In later stages, added complexity will arise from scattered remnants of old ecosystems that are left behind as ecosystems move northward and upslope.

Turning finally to species, the level of threat posed by climate change hinges on the balance between adaptive capacity and the amount of anticipated warming. Several insights into this dynamic can be gleaned from the Holocene paleoecological record (Moritz and Agudo 2013). Canadian species are all Ice Age survivors, having experienced a 4–5°C change in global temperature during the Holocene (Clark et al. 2016). The paleoecological record shows that all of our species, including those with low apparent dispersal ability, were able to shift ranges across vast distances. In some cases, Canadian species took refuge as far south as the southern US (Pielou 1991). It is also apparent that all current species, including habitat specialists and endemic species, have the flexibility to persist under novel conditions. Glacial refugia were not replicas of current ranges. They were composed of different species assemblages, and there were differences in photoperiod, soils, and other site-related factors (Pielou 1991).

An important feature of the present episode of warming is that the rate of change appears to be faster than in previous episodes. Many species may be unable to adjust their range quickly enough to track their preferred conditions, subjecting them to climatic disequilibrium (Malcolm et al. 2002; Corlett and Westcott 2013). The ability of species to persist in a state of climate disequilibrium is currently a topic of active debate. Some researchers assert that climate envelope fidelity is necessary for species persistence, and their predictions concerning species viability are often dire. For example, a highly cited paper by Thomas et al. (2004) predicts that 15–37% of global endemic species will go extinct because their current climate envelope will disappear or become inaccessible to them.

The counterargument is that climatic disequilibrium is not a novel phenomenon. It was also experienced during the Ice Age transitions. Global temperatures had already risen to near current levels by the beginning of the Holocene (11,700 years ago; Clark 2016), but returning species could not immediately establish themselves in their current ranges. Much of Canada was still under ice at that time, and freshly exposed landscapes lacked soil (Pielou 1991). Despite the climatic and ecological disequilibrium that species must have experienced during this period, extinction rates did not increase (McInerney and Wing 2011; Willis and MacDonald 2011). Based on accumulating paleoecological evidence, some researchers now believe that species have greater climatic flexibility than has previously been assumed (Pearson 2006; Hof et al. 2011).

Even if climate envelope fidelity is not an absolute requirement for persistence, there undoubtedly are limits to how much accommodation is possible. We do not know what the thresholds are, but the greater the degree of disequilibrium, and the longer it lasts, the higher the likelihood that some species will decline or become extirpated as a result. It is also evident that certain species traits increase vulnerability (Pearson et al. 2014):

- Narrow physiological tolerance limits
- High degree of habitat specialization
- Small range size
- Low dispersal ability

- Low rate of reproduction
- Low genetic variation within and among populations
- Sensitivity to human disturbance

A caveat when using Ice Age comparisons is that the Quaternary featured a change from cold to warm, whereas we are now changing from warm to hot. This has particular relevance for cold-adapted species found in high alpine areas and along the Arctic coast. These species are likely to be confronted with conditions that are novel to them, and which cannot be accommodated through range shifts. Pushed toward mountaintops and the sea, some of these species may find themselves trapped—physically unable to shift to cooler sites and outcompeted by species adapted to warmer climates.

Another important caveat is that past warming episodes occurred in the absence of an anthropogenic environmental footprint. More than anything else, it is the synergy between environmental degradation and climate change that raises the spectre of widespread species extinction (Hof et al. 2011). Species that are sensitive to human disturbance have experienced range contractions and declines in abundance and are generally less resilient to change of any type. For those species currently struggling to remain viable, the added stress imposed by rapidly changing climatic conditions may be what ultimately seals their fate. More generally, physical barriers and habitat deterioration can reduce adaptive capacity by hindering species movements and range adjustments (Nixon et al. 2016).

In summary, climate change will have both positive and negative influences on biodiversity. Some species and ecosystems will undoubtedly decline, as their climate envelopes contract. But overall, the moderating of Canada's harsh climate, together with complex transitional dynamics, will promote an increase in diversity across the country. The caveat is that these positive outcomes may not be realized if human land uses preclude effective adaptation. The proportion of species "left behind" may be relatively small under low-end warming scenarios, assuming that management efforts are made to reduce barriers to movement and to actively assist highly vulnerable species. But widespread adaptation failure is a real possibility under high-end warming scenarios, which take us into uncharted territory.

The Foundations of Climate-Ready Conservation

Dynamic Baselines

The fundamental goal of conservation is to protect natural systems from the deleterious effects of human activities (see Chapter 2). This remains unchanged under climate change. Indeed, it is imperative that we do not lose sight of this goal (Maxwell et al. 2016). However, the historical baselines that have generally been used to define the natural state are no longer tenable. As temperatures increase, ecosystems will adjust accordingly, which means there will be no going back to the way things were in the past, even in principle. An ecological baseline that incorporates climate change is needed instead. Establishing this baseline requires us to revisit the concept of "natural" that we developed in Chapter 7.

On the one hand, there is a clear scientific consensus that the present episode of warming is anthropogenic in origin, resulting from our release of CO_2 and other greenhouse gases (Cook et al. 2016). Because climate change is not natural (in this instance), it should logically be considered an anthropogenic threat that needs to be countered. On the other hand, once CO_2 and other greenhouse gases are released, they become indistinguishable components of the atmosphere. The threat they pose is amorphous and global in scope, beyond the purview of local conservation managers. Moreover, the process is irreversible on timescales relevant to biodiversity management (Archer and Brovkin 2008). As a global society we can (in principle) control how much CO_2 we release, but we have no control over changes in climate that arise from the CO_2 already added to the atmosphere.

The pragmatic solution is to deal with climate change on two levels: **climate mitigation** and **climate adaptation**. The idea is to differentiate the release of greenhouse gases from the climatic and ecological effects they produce. Climate mitigation refers to preventative efforts, most of which are focused on reducing the release of greenhouse gases (Hansen et al. 2013). These efforts need to be global in scope to be effective. It does not matter if the CO_2 comes out of the tailpipe of a car in Hamilton or the smokestack of a power plant in China—it all pools together. This requires international government cooperation and policies that influence the actions of businesses and individuals.

Currently, the main international effort to curb greenhouse gas emissions is the Paris Climate Agreement, to which Canada is a signatory. This agreement sets national targets for reducing greenhouse gas emissions, with the aim of keeping the global rise in temperature below 2°C (UN 2015). Unfortunately, global emission reductions to date have been uneven and collectively fall substantially short of what is required to achieve the Paris Agreement goal (Raftery et al. 2017).

The climatic changes that result from greenhouse gas emissions demand a different response. What is needed is adaptation rather than mitigation. Climatic changes cannot be reversed, and efforts to prevent ecological systems from responding to changing conditions would be counterproductive and ultimately futile (Dunlop et al. 2013; Hamann and Aitken 2013). Biodiversity is best served by treating all forms of climatic change as natural phenomena and helping species adapt to these changes.

Because the climate is changing, the ecological baselines we use to define working objectives for conservation need to become dynamic, describing the natural *trajectory* of change rather than the natural historical *state* (Simberloff 2015). The natural trajectory is what we would observe within a large, pristine protected area over time (Murcia et al. 2014). Indeed, monitoring the climate-induced transitions that occur within protected areas is one of the approaches that can be used to characterize dynamic baselines.

In summary, climate change presents a threat to biodiversity that must be addressed proactively, through the control of emissions, rather than after the fact. Climatic changes that occur despite preventative efforts are essentially irreversible and should be accommodated rather than resisted. This can be accomplished by defining a dynamic ecological baseline and focusing on facilitating climate adaptation, rather than resisting ecological change. The aim is to ensure that climate-induced ecological transitions unfold as they would in undisturbed systems. For the most part, this means attending to conventional threats using conventional methods. We will examine dynamic benchmarks and related conservation methods in more detail later in the chapter, in the context of specific conservation applications.

A remaining challenge with climate adaptation involves the social dimension of conservation. Unmoored from the perceived objectivity of the preindustrial landscape as a benchmark, support for conservation may waver. Some voices are already beginning to question whether any systems can be considered natural, and thus worth conserving, in a world of constant change (Murcia et al. 2014). Harris et al. (2006) write:

> We must tread very carefully. A consequence of rapid climate change may be the loss of public interest in conservation and restoration goals. Inured to the change, the idea of supporting painstaking restoration goals will give way to functional, emergent, and designer ecosystems. (p. 175)

A **designer ecosystem** is a system engineered to provide specific ecosystem services (Higgs 2017). The concept arose in the context of restoring highly degraded sites, but it is now being discussed in the context of the novel ecosystems produced by climate change (Higgs 2017; Backstrom et al. 2018). This raises serious concerns, as expressed by Murcia et al. (2014):

> What is at stake is whether we decide to protect, maintain, and restore ecosystems wherever possible or else adopt a different overall strategy, driven by a vision of a 'domesticated' earth, and use a hubristic, managerial mindset. (p. 552)

While it is true we can no longer keep natural systems exactly as they were in the past, this is not grounds for abandoning the fundamental tenets of conservation. The value of biodiversity does not diminish in a warmer world, and protecting species from the deleterious effects of human activities is no less important. After all, species are not changing, just their location. Conservation practitioners need to help the public and decision makers understand these distinctions and ensure that fundamental concepts about biodiversity and conservation are not abandoned.

Climate Scenarios

Another step in adapting conservation to climate change is incorporating climate projections into the planning processes we have discussed in previous chapters. Conservation practitioners should have a basic understanding of how these projections are created and how they can be obtained and utilized. It is also important to understand their limitations.

The process begins with projections of future greenhouse gas emissions. These emissions depend on global population growth, economic growth, land use, technological innovations, and most importantly, social awareness, concern, and willingness to respond (Van Vuuren et al. 2011). Because of the wide range of possibilities, emission trajectories are not amenable to quantitative modelling. Instead, the Intergovernmental Panel on Climate Change has developed a suite of four **emission scenarios**, termed Representative Concentration Pathways (Van Vuuren et al. 2011). As discussed in Chapter 7, scenarios allow us to explore plausible alternative futures without committing to them as forecasts (West et al. 2009).

The second stage in projecting the future climate involves simulating global climatic processes on the basis of fundamental physical principles. These climate models, formally referred to as **general circulation models**, have been developed by several teams around the globe. By using the same set of emission scenarios as inputs, comparisons can be made among models and among emission scenarios.

Climate models are evolving rapidly, but many climatic processes, especially those involving feedback loops, are still only partially understood. As a result, the climate models developed by the various modelling teams differ in important ways and produce different results under the same emission scenarios. This variance among models is an expression of modelling uncertainty. As a rule, the farther into the future that projections are made, the higher the level of uncertainty. The year 2100 is used as the limit for most management applications.

The climate projections from more than 20 international models are readily available to conservation practitioners (Wang et al. 2016b). It is not data acquisition that presents a challenge, but data overload. Working with the temporal projections from all available models and all four emission scenarios is not practical.

The obvious solution is to focus on the most reliable models; however, reliability is not easily determined. The available comparative studies tend to focus on specific climatic processes, such as the simulation of clouds, rather than overall performance (Jiang et al. 2012). In any case, there is really no gold standard to test against. Examining how well the models replicate past climatic patterns is helpful, but it is not a dependable guide to their reliability in future periods when CO_2 concentrations will be much different.

A better approach is to treat the entire gamut of climate projections as scenarios, rather than predictions. We can then select a subset of these scenarios to represent the full spectrum of potential climate outcomes. This is illustrated in Fig. 9.8, which shows the temperature and precipitation projections to 2080 under high- and low-emission scenarios as predicted by a suite of climate models. The four peripheral circles are candidates for representing extreme scenarios (i.e., coolest, hottest, wettest, and driest). The central circle would be an appropriate choice to represent an intermediate climate scenario. The coolest scenario has special relevance for management because it carries a high level of certainty—there is complete agreement among all models that the climate will become *at least* this warm.

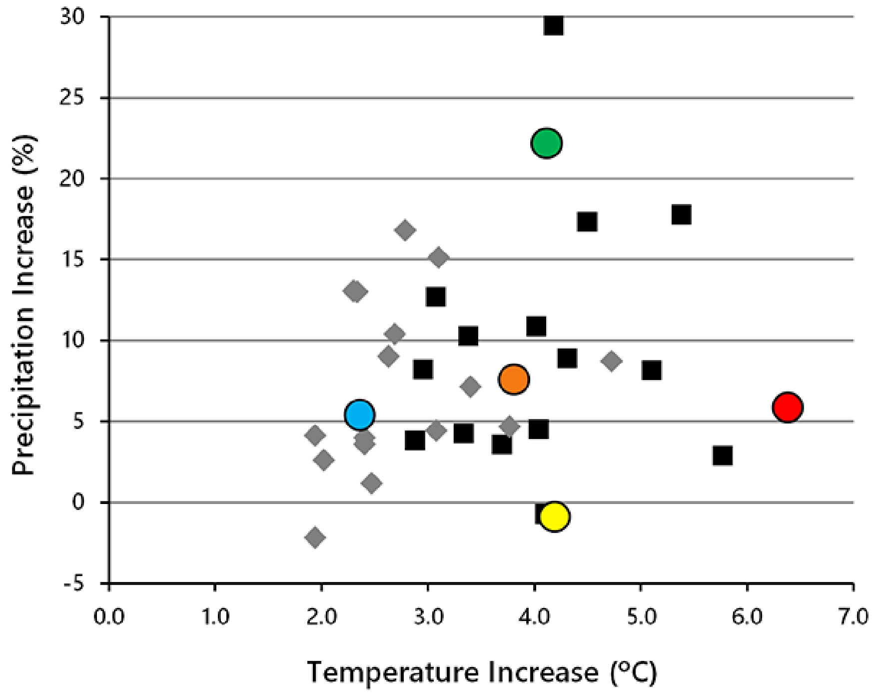


Fig. 9.8. The projected increase in mean annual temperature and precipitation to 2080 for 18 climate models. The grey diamonds represent a low-emission scenario, and the black squares represent a high-emission scenario. The coloured circles are candidate climate scenarios for use in planning applications. Data are for the province of Alberta, adapted from Schneider 2013.

An alternative approach is to pool the projections from all models into an ensemble mean. This approach is less complicated but provides no insight into the range of outcomes possible. Consequently, it is not appropriate for planning horizons beyond mid-century, when climate projections begin to diverge significantly.

The last step is to link the climate projections to the models used to support management decisions. Many of these models include climate-sensitive parameters that can be dynamically adjusted on the basis of the projected future climate. For example, Battin et al. (2007) incorporated climate change into a population model for Chinook salmon via parameters for water temperature and stream flow. This enabled them to explore the effects of warming on habitat restoration efforts (Fig. 9.9). Other climate-sensitive parameters commonly used in decision support models include winter survival, soil moisture, and the rate of natural disturbances.

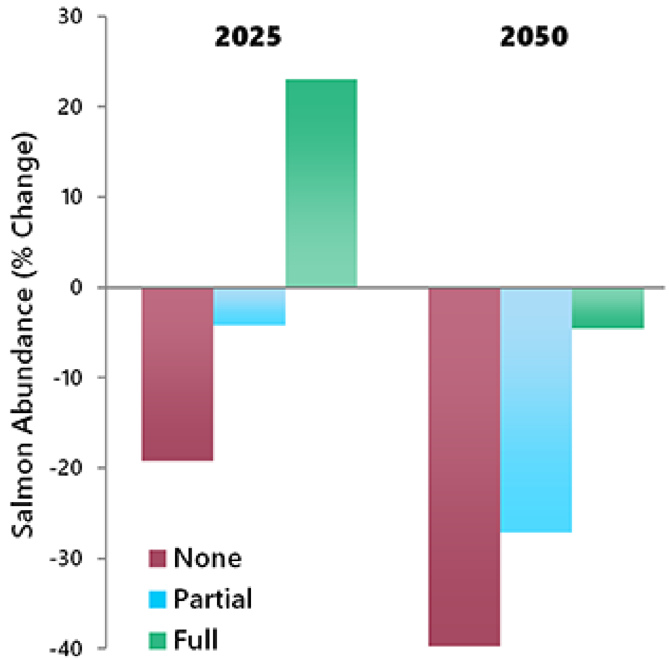


Fig. 9.9. The predicted effects of climate change on habitat restoration efforts for Chinook salmon in the Pacific Northwest. The first group of three bars illustrates the expected abundance of salmon in 2025 under three levels of restoration. Only full restoration results in an increase in abundance. Under the warmer climate in 2050, shown in the second set of bars, none of the restoration efforts is able to maintain salmon abundance. Adapted from Battin et al. 2007.

Climate projections can also be used in combination with bioclimatic envelope models to predict changes in species distributions and changes in habitat conditions, as previously discussed. This information can be integrated into decision support models or used to directly inform decisions. The main caveat is that the bioclimatic envelope projections represent equilibrium conditions and do not account for the time needed to achieve these conditions. Furthermore, our ability to predict changes under the hottest scenarios is limited. This is uncharted territory, and our statistical models may break down under such conditions.

Robust Decision Making

Having discussed dynamic baselines and the incorporation of climate change into decision support models, we now turn to the decision-making process itself. Because the baseline is no longer fixed, greater consideration has to be given to the long-term repercussions of conservation actions. The best course of action in the near term may not be optimal over the longer term, setting up the potential for trade-offs among time periods.

Planning efforts must also grapple with the added uncertainty that climate change presents. In conventional planning, we use models or expert opinion to forecast outcomes under alternative management approaches, and then select the option that best achieves the stated objectives. It is understood that these forecasts are subject to uncertainty, but we assume they are reliable enough to differentiate the performance of the management alternatives under consideration. To backstop this assumption, efforts are made to identify points of uncertainty and to address these uncertainties through additional research. In addition, the state of the system is monitored over time, and variances between the plan and actual outcomes are addressed through periodic replanning.

For short-term planning (i.e., time horizons under ~20 years), conventional planning approaches remain viable, despite the added uncertainty from climate change. Climate model projections are reasonably consistent at this early stage (Fig. 9.10). Moreover, the amount of warming is relatively subdued, compared with later periods, and is unlikely to result in unpredictable ecological outcomes.

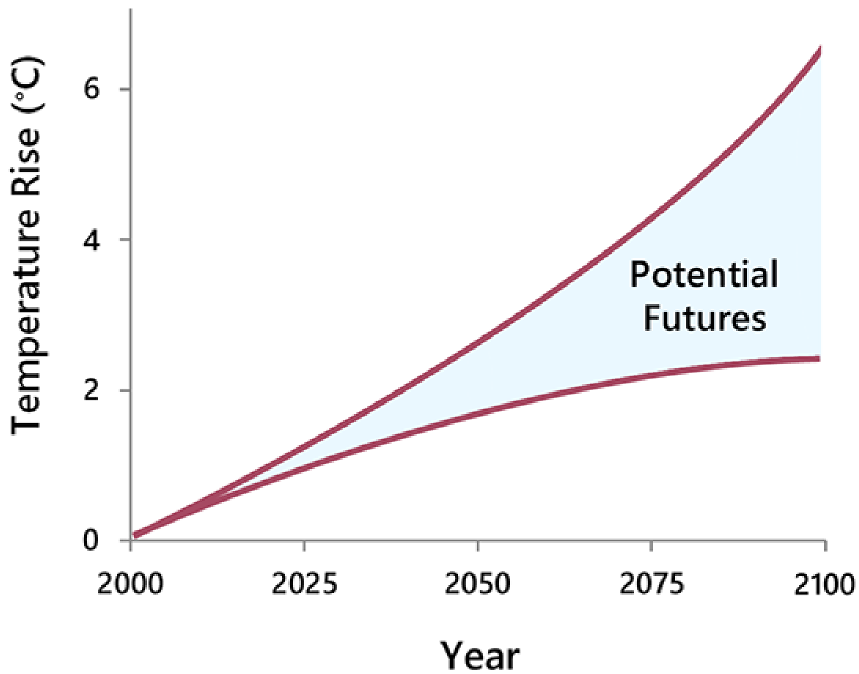


Fig. 9.10. The range of potential future climates diverges significantly in the latter half of the century because emission scenarios become more distinct and climate modeling uncertainties increase. Adapted from Schneider 2013.

Climate change presents a much greater challenge for long-term planning. The range of potential future climates expands significantly in the latter half of the century (Fig. 9.10), forcing us to reconsider the meaning of optimality. The performance of management approaches—and hence our assessment of which is best—is unlikely to be the same under distinctly different climates (Fig. 9.11). The relative rankings may change under different climate scenarios, and there may be no management approach that is consistently optimal under all conditions.

For such situations, a "**robust**" or "**no regrets**" approach to decision making may be most appropriate (Millar et al. 2007; Kunreuther et al. 2013). The basic idea is to select a management option by how well it performs across *all* potential futures, rather than just one. The simplest method is to select the management option with the highest mean score across all options (this would be Option C in Fig. 9.11). Alternatively, priority might be given to the management option that has the least variance or the best worst-case outcome (Option B in Fig. 9.11). There are also more sophisticated mathematical approaches, such as the minimax-regret method, that permit the preferential weighting of scenarios by their perceived likelihood (Kunreuther et al. 2013).

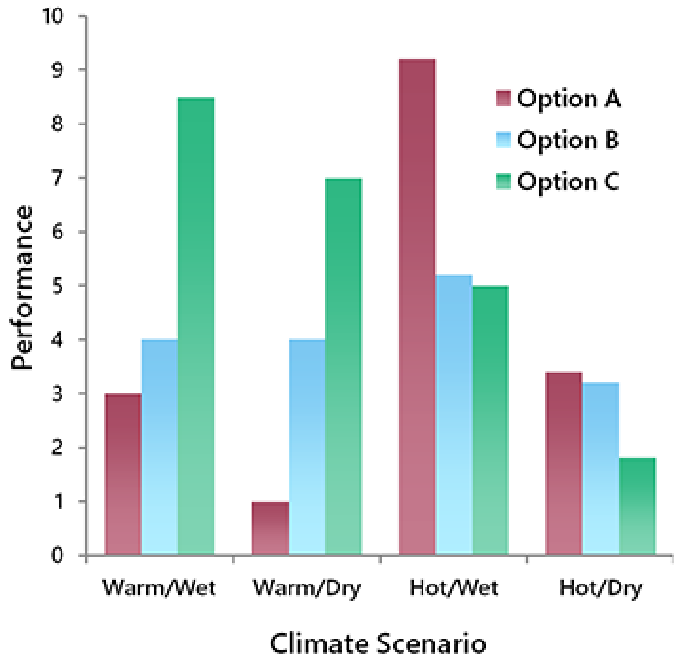


Fig. 9.11. A hypothetical example illustrating the long-term performance of three management options under four climate scenarios. Option A achieves the highest performance, but only if the climate is hot and wet. Option B provides the most consistent performance. Option C delivers the best mean performance.

Conservation approaches that perform well across varying conditions come in various forms. One approach is to enhance a system's overall resilience (Seidl 2014). Efforts to reduce the intensity of industrial impacts and to limit cumulative effects fall into this category.

Another way of achieving robust performance is **bet-hedging** (Millar et al. 2007). Bet-hedging addresses deep uncertainty by simultaneously applying different strategies across the landscape. This is basically a risk-spreading strategy, which is useful for avoiding widespread management failure when the outcome of potential strategies can not be predicted in advance. The bet-hedging approach can also serve as an effective method of increasing knowledge, especially for new approaches that have not yet been adequately field tested. There is overlap here with the concept of adaptive management, which we will discuss in Chapter 10.

A variant of bet hedging is the **optimal portfolio approach** (Crowe and Parker 2008; Ando and Mallory 2012). In this case, instead of applying different actions in different places, a single multipronged strategy is applied throughout the planning area. For example, in the context of reforestation, diverse seed stock might be used to maximize genetic and species diversity across the landscape, in the hope that some genotypes and species will thrive regardless of how the climate changes.

Institutional Support

Adaptation to climate change can be enhanced or hindered by institutional structures and norms (Williamson et al. 2012). Though there is now widespread awareness of the need for adaptation, there is still great uncertainty about what should be done and there are various barriers to implementation (see Box 9.2). Initial efforts have focused on information gathering and dissemination, vulnerability assessments, and research.

Structural changes to decision-making systems will eventually be needed. These are only now being contemplated and will take time to be realized. The challenge is to develop a system that embraces flexibility while safeguarding against activities that are inconsistent with the aims of conservation and abuse by actors seeking to avoid environmental regulation (Craig 2010). Furthermore, while we must accept that management outcomes are less predictable under climate change, companies and government agencies must still be held accountable for the decisions they make and the actions they take (Hagerman et al. 2010a). It is as yet unclear how this might be accomplished.

Because climate change is occurring at scales much larger than even the largest planning regions, successful adaptation will require collaboration among jurisdictions (Heller and Zavaleta 2009; West et al. 2009). Additional funding and staff will also be needed. At present, managers wishing to implement adaptation programs usually have to scavenge funds from other programs, which is not a viable long-term solution. Finally, there is a need for enhanced monitoring and additional research. Pilot projects are a promising approach, serving as laboratories for identifying and solving the many practical issues that must be addressed.

Box 9.2. Climate Adaptation in the Slow Lane

Despite widespread attention to climate change, and discussions about what should be done, little demonstrable change is evident at the operational level of conservation (Poiani et al. 2011). There are various reasons for this and they must be understood and addressed if climate adaptation is to be widely implemented (Magness et al. 2012; Hagerman and Satterfield 2013; Lonsdale et al. 2017). The barriers include:

- **Scientific uncertainty.** Although little doubt remains about the overall trajectory of climate change, there is considerable uncertainty about the amount and rate of change. There is also uncertainty about how species and ecological systems will respond. Some decision makers may hesitate to act until empirical evidence of ecosystem changes validates model predictions.
- **Capacity limitations.** Because of capacity limitations, managers often must focus on the most pressing issues and are unable to provide much attention to slowly evolving issues like climate change. A shortage of relevant technical expertise is also an issue.
- **Resistance to change.** Human beings have a natural tendency to resist change. In the context of climate change, individuals that have dedicated their careers to the current system of biodiversity conservation may refuse to accept that different approaches are needed. Within the public sphere, there may be skepticism about government and industry motives.
- **Lack of an alternative.** Knowing about an issue does not lead to immediate change. Before the status quo can be abandoned, a viable alternative must be available. In the case of climate adaptation, the development of alternative approaches is still at an early stage.
- **Political inertia.** Past trade-off decisions concerning land use often involved hard-fought battles between opposing interests. Reopening these decisions to accommodate climate change has com-

plex political ramifications. The environmental community may resist change, not because it is opposed to climate adaptation per se, but because it is concerned that industry will be given a free hand under the guise of increased flexibility. For their part, managers may hesitate to implement novel management approaches because of a political culture that is not accepting of failure.

Ecosystem-Level Conservation

We now turn to conservation practices, beginning with ecosystem-level approaches. The fundamental goal of conservation does not change under climate change—it is still to protect biotic systems from the deleterious effects of human activities. Therefore, the conservation tools we discussed in Chapters 6–8 remain relevant and are applied largely for the same purposes. The main change, in terms of working objectives, is that we must now also ensure that climate-induced ecological transitions unfold as they would in undisturbed systems. This entails facilitating species adaptation.

There are four main concerns related to adaptation that warrant management intervention:

1. The natural capacity for adaptation of many species has been compromised by human activities, making them less able to withstand climatic variability and less able to shift their range

2. Barriers to movement now exist, including regions where habitat quality has deteriorated

3. This episode of warming is particularly rapid and may exceed the intrinsic adaptive capacity of some species, especially if the warmest scenarios are realized

4. Non-native species may competitively exclude the movement of native species into new areas

In terms of implementation, adapting conservation to a changing climate is mainly a matter of fitting existing conservation methods into a dynamic framework (Lindenmayer et al. 2010; Abrahms et al. 2017). No truly novel methods exist for addressing climate change, though some existing approaches may be used in new ways.

Protected Areas

The need to provide species with a refuge from anthropogenic disturbances does not diminish under climate change. In fact, the need is greater. Protected areas increase the resilience and adaptive capacity of species, thereby supporting climate adaptation (Hodgson et al. 2009). In addition, protected areas enhance landscape connectivity and serve as ecological benchmarks—roles that take on added importance under climate change.

Because the distribution of species and ecosystems is destined to shift, reserve planning now requires a dynamic framework (Game et al. 2011). Rather than trying to protect specific assemblages in specific locations, we should aim to provide species with continual access to protection as their range shifts across the landscape. This is best accomplished by representing the stable geophysical attributes that underlie ecological patterns (Anderson and Ferree 2010; Groves et al. 2012). Simply put, our objective should be to represent the major "arenas" of biological diversity, not the "actors" (Beier and Brost 2010). Bioclimatic modelling suggests that this approach should succeed in maintaining coarse-filter representation as long as movement among reserves is not overly restricted (Schneider and Bayne 2015).

A key challenge in implementing the arenas concept is identifying the arenas (Schneider and Bayne 2015). What is the appropriate scale? Which geophysical attributes should be used? Are some attributes more important than others? And how do we delineate features that blend seamlessly from one to the next? These are difficult questions to answer from a purely abiotic perspective. Therefore, we must look to ecological patterns for guidance. The geophysical features that matter are those that have a demonstrable ecological effect.

A hybrid system, such as the National Ecological Framework, is well suited to this task (Fig. 9.12). This classification is fundamentally based on broad geophysical features and patterns. However, vegetation patterns are incorporated as well, providing insight into the ecological relevance of the observed geophysical features and indicating where distinctions are warranted. There is no guarantee that the ecosystem boundaries in this classification will all remain stable under climate change. But this approach should be robust enough to ensure that the major biotic arenas have been represented.

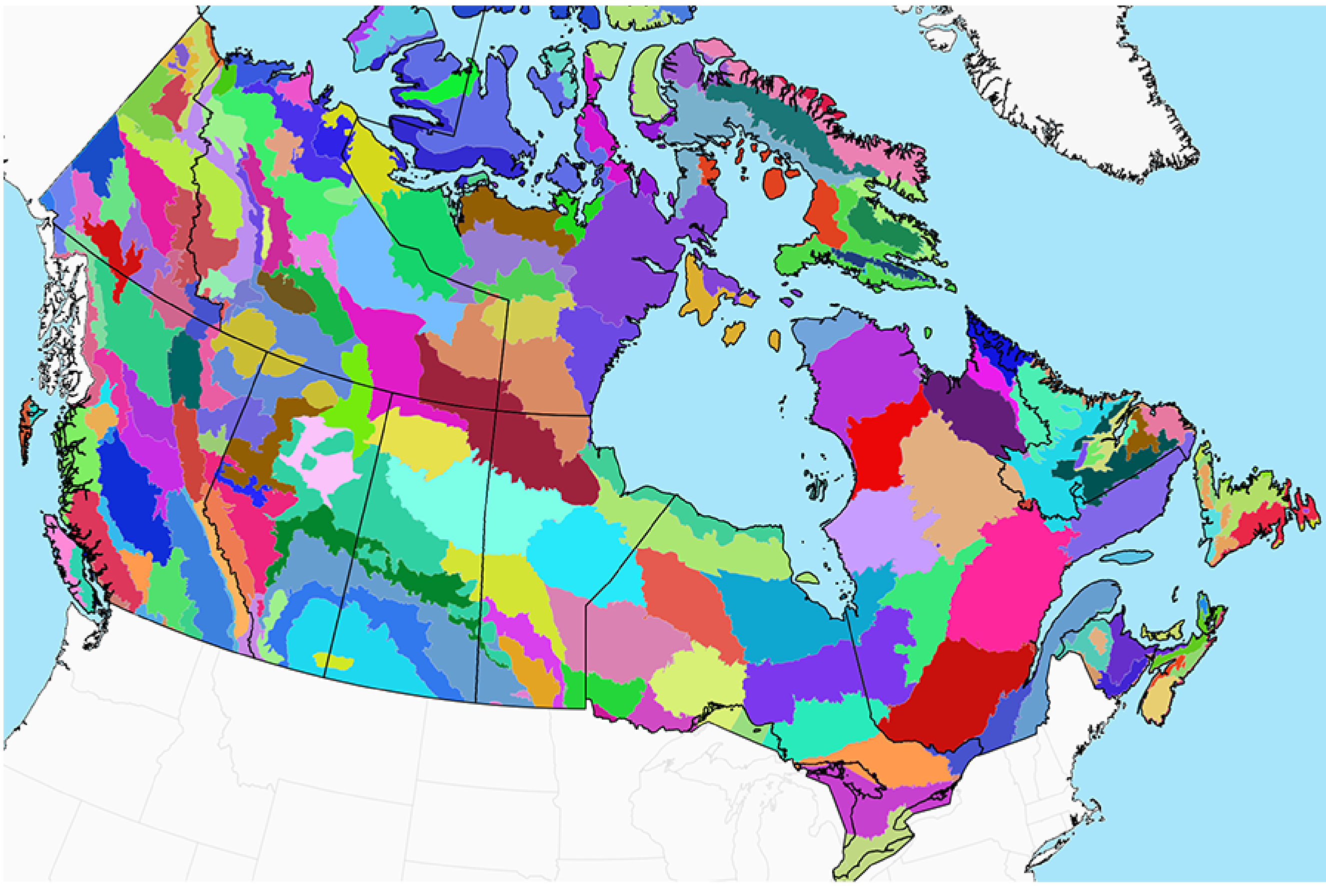


Fig. 9.12. The ecoregions of Canada, as defined by the National Ecological Framework. Ecoregions are one level below the ecozones shown in Fig. 1.3.

Finer-scale landscape features can be represented through "**land facets**," which are landscape units with uniform topographic, geologic, and soil attributes (Wessels et al. 1999; Beier and Brost 2010). For example, land facets might include marshes, patches of sandy soil, riparian zones, and other features relevant to the region. In contrast to the large polygons of the National Ecological Framework, which are all unique and merit individual representation, land facets are *recurring* landscape features. The intent is not to represent each facet individually, but to capture a sample of each type within each ecoregion.

At present, there is no established classification system for land facets in Canada. Therefore, the task of defining them falls to protected area planning teams. As previously noted, it can be challenging to determine which features to include and how to set breakpoints. As a general rule, features should be selected because of their ecological importance, not just because the GIS layers happen to be available.

Climate Refugia

A new dimension to reserve design that has emerged in response to climate change is the protection of climate refugia, of which there are various types (Ashcroft et al. 2012; Morelli et al. 2016). One type, termed **in situ refugia**, involves landscapes that are expected to be relatively resistant to ecological change (Rose and Burton 2009). For example, a 2°C rise in temperature in a mountainous area represents only a few hundred metres of vertical rise (Loarie et al. 2009). A single vertically oriented reserve in such an area could serve as a climate refugium because it could maintain representation of most pre-existing ecosystem types, even if they shift upslope under moderate levels of warming. In contrast, a 2°C rise in temperature in a region with little topographic relief may generate a northward displacement of several hundred kilometres, precluding comprehensive protection.

The northern fringe of large ecosystems may also serve as in situ refugia, since these areas will be among the last to transition. However, such refugia should not be considered permanent, even if bioclimatic projections suggest they will remain stable. Climate change will continue until well into the next century, beyond the scope of our models. Few, if any, ecosystem remnants are expected to remain in place indefinitely. What we are identifying are areas of *relative* stability, which is still important for conservation. Protecting these areas may provide species with the time they need to adjust to the changing climate.

As with other reserve design elements, the effects of including in situ refugia in a reserve network should be explored with conservation planning software. Trade-offs with other design objectives can be expected. Because refugia are based on climate model projections, the level of uncertainty is high, and this will weigh against them in trade-off analyses

Ex situ climate refugia are also possible. The idea here is to protect the projected *future* location of selected ecosystem types. This might be proposed for ecosystems that have high ecological or social value, or that are expected to become rare in the future. This approach is technically feasible, but the selection of such sites may be difficult to defend. Why should some ecosystem types be afforded ex situ protection over others? Which point in time should be targeted? How much uncertainty about future projections is acceptable? These questions are difficult to answer objectively.

Climate refugia can also be identified at scales finer than those normally used for protected area planning. These **micro-refugia** are local sites of ecological stability that persist, at least for a time, after the regional climate envelope has transitioned to something new (Dobrowski 2011; Lenoir et al. 2017). Their existence is a consequence of topographic complexity and associated fine-scale climatic patterns. Because micro-refugia are small and diffuse, they do not lend themselves to protection through conventional reserves. However, their temporary protection through special management zones may provide many species the extra time they need for climate adaptation.

Connectivity

Climate change greatly increases the need for landscape connectivity, to support the ability of animals and plants to track their preferred climatic conditions at the necessary pace. The management tools for enhancing connectivity described in earlier chapters (i.e., generic connectivity measures, movement corridors, and protected areas) remain the mainstay approaches under climate change. No new options are available.

Generic connectivity measures facilitate movement in all directions and are therefore robust under climate change. Because they apply to the entire land base, these measures are well suited to facilitating range shifts and enhancing the viability and adaptive capacity of species (Doerr et al. 2011). The only change needed to accommodate climate change is a redoubling of efforts.

Corridors facilitate movement in specific directions, so their effectiveness is affected by the directionality of movement imposed by climate change (Littlefield et al. 2017). This needs to be accounted for at the design stage (Beier 2012; Nunez et al. 2013). For most species, range shifts will occur along existing climatic gradients, typically northward or upslope. Planners should assess corridor options on the basis of their alignment with such gradients, favouring those that are most aligned (Rouget et al. 2006; Nunez et al. 2013).

Protected areas present a special case in that they contribute to landscape connectivity while also depending on connectivity to remain functionally linked. Both aspects need to be considered in the reserve design process, with special attention given to the effects of climate change.

Connectivity among reserves can be maximized at the planning stage by taking advantage of the directionality that climate change imposes on species range shifts. For example, instead of creating a circular reserve, it would be advantageous to create an oblong reserve oriented in the direction that climate envelopes are expected to move. Connectivity can also be enhanced across the entire reserve network by arranging large reserves and smaller "stepping stone" reserves in a configuration that minimizes the distance between reserves along a relevant climatic gradient (Fig. 9.13; Robillard et al. 2015).

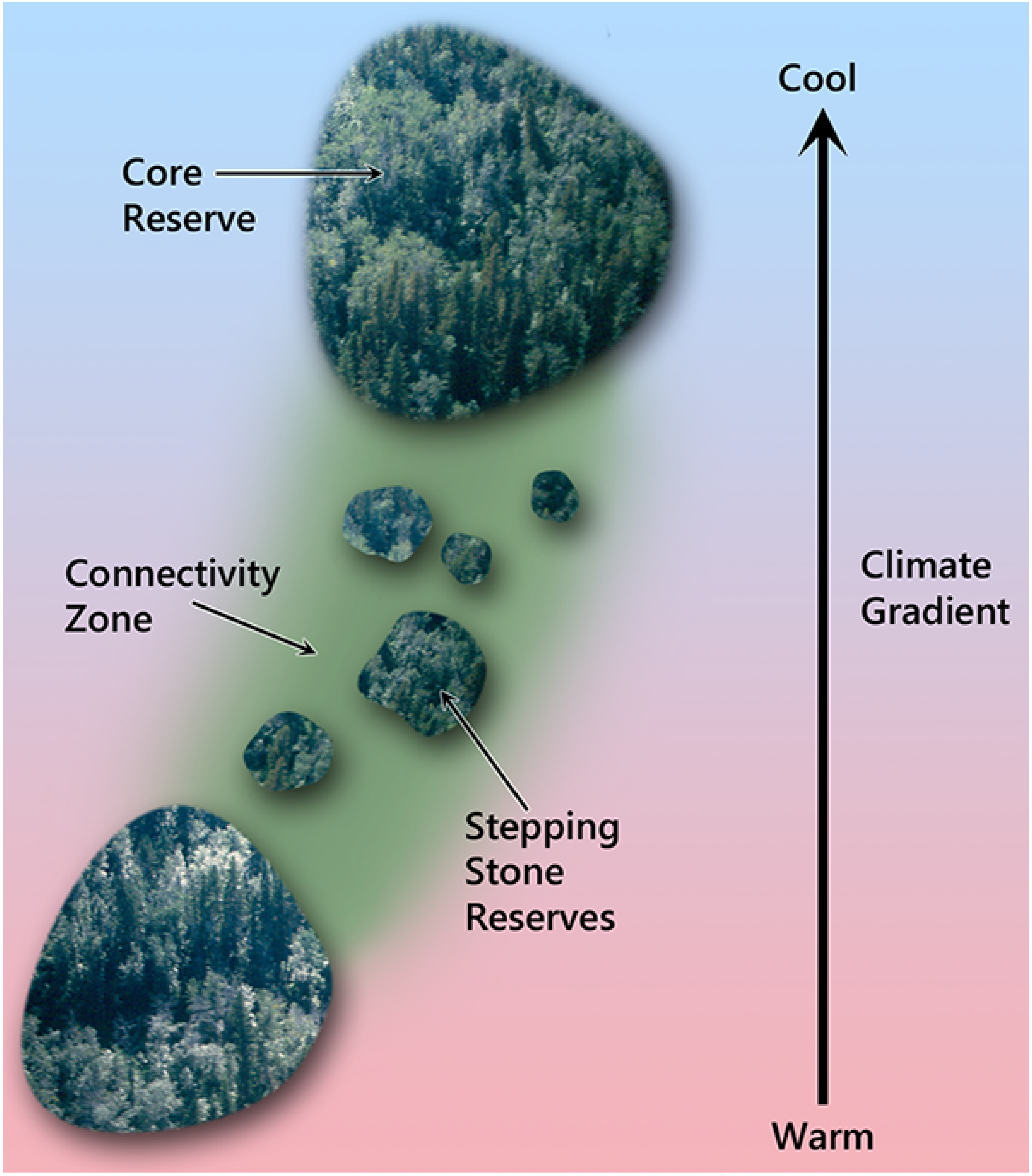


Fig. 9.13. Long-distance range shifts can be facilitated by aligning large core reserves, stepping stone reserves, and special management zones with the directionality imposed by climate change.

Design approaches for enhancing connectivity among reserves are subject to several practical limitations. In particular, we are not working with a blank slate; more than 12% of Canada has already been protected. This places constraints on the overall reserve system configuration, limiting options for minimizing distances along the axes preferred for accommodating climate change. Also, the incorporation of preferred spatial axes is not something that conservation planning software was designed to do. Time-consuming work-arounds and manual approaches are required. Finally, the distances between existing reserves are often vast and not easily bridged.

The placement of new reserves along preferred climatic axes may also conflict with objectives related to representation, since the respective spatial priorities are unlikely to align just by chance. When conflict is unavoidable, planners should try to find a workable balance, rather than emphasize one class of design objectives over another. For example, it may be necessary to accept gaps in fine-scale representation to obtain the flexibility needed to achieve functional objectives like connectivity.

The Natural Disturbance Model

The natural disturbance model is a method for reducing proximate threats to biodiversity from industrial disturbances on the working land base. It remains an important conservation tool under climate change, helping to prevent species declines and enhancing resilience and adaptive capacity.

The natural disturbance model is not intrinsically forward-looking, so its application does not depend on climate projections or bioclimatic modelling. The intent is simply to modify industrial practices such that ecological composition, structure, and processes remain within the natural range of variation (NRV). Estimates of NRV that are based on historical conditions remain acceptable for now because climate-induced ecological transitions are not yet widespread (Gauthier et al. 2023). In the future, a dynamic ecological baseline will be needed.

A dynamic version of NRV can be obtained by monitoring relatively undisturbed landscapes and then extrapolating the findings to the working landscape. As undisturbed sites evolve under warming temperatures, so too will NRV. The larger sites in our existing system of protected areas are obvious choices as benchmarks; indeed, this is already one of their central roles (Wiersma 2005). However, significant gaps in representation exist in many regions. To some extent, these gaps can be addressed by monitoring smaller natural areas and sites that are only partially protected. But it would be best to fill gaps in representation by adding new protected areas, adding impetus to Canada's commitment to protect 30% of its landscapes. To maximize their effectiveness as benchmarks, the new sites should be large enough to maintain and represent natural ecological processes.

Box 9.3. A Functional Approach for Defining Ecological Baselines

To accommodate climate change, some researchers have proposed that ecological baselines should be based on ecosystem function, rather than specific structural and compositional attributes (Hagerman et al. 2010b; Glick et al. 2011). For example, it has been suggested that we maintain "functional" ecosystems and ecosystem "health." The appeal of these sorts of measures is that they obviate the need for periodically updating the ecological reference state.

The problem with this approach is that objectives like maintaining ecosystem function and health are inherently ambiguous and malleable. We achieve robustness at the expense of clarity. Furthermore, the relationship between ecosystem function and species diversity is quite tenuous (Srivastava and Vellend 2005). The loss of individual species, especially those that are rare, is unlikely to be detected by any of the available measures of ecosystem function (Schwartz et al. 2000).

Given the malleability of functional objectives and the lack of an explicit connection to species diversity, ecological baselines defined solely on the basis of functional attributes provide a poor foundation for biodiversity conservation. The functional approach is best restricted to ecosystem services applications. For conservation applications, an integrated approach that includes structure, composition, *and* specified functions, explicitly linked to natural systems, is best.

Restoration and Reclamation

Like the other ecosystem-level methods we have discussed, restoration and reclamation also need to be placed in a dynamic framework. Since most of these projects are focused on the restoration of vegetation, the main issue is determining the appropriate source of seed stock.

One line of reasoning suggests that seed stock should be obtained from warmer sites along a relevant climatic gradient. The idea is to preadapt restored sites to future climates to maximize their long-term stability (Harris et al. 2006). This approach has gained significant support in the forestry sector. Regulations requiring the use of local seed stock in reforestation programs are now being modified to permit the use of seed from warmer climatic zones (Klenk 2015).

An alternative view is that we should continue using local sites as seed sources. In this case, the conservation of local genetic adaptations is the overriding concern, not the climate-proofing of restored sites (Gibson et al. 2009). Although local populations may not be well suited to future climates at their current location, their genetic adaptations will be important in facilitating range shifts into cooler landscapes in future decades (Gibson et al. 2009; Hamann and Aitken 2013).

Both of these perspectives have merit and the choice of which seed source to use depends on the project objectives and local context. For example, restoration projects in agricultural landscapes may emphasize local seed sources, given the importance of conserving local genotypes when little native vegetation remains. Conversely, reclamation projects in northern forests may emphasize preadaptation, given the slow growth and longevity of trees. A proviso when using stock from warmer sites is that the stock must be sufficiently winter hardy to survive in the restoration site under current climate conditions. There are also regulatory issues that must be addressed if the seed stock is coming from the US.

Invasive Species

Climate change will make Canada's harsh climate more hospitable to alien species, adding a new dimension to invasive species control (Smith et al. 2012). Our ability to achieve successful outcomes will decline, and costs will rise, suggesting that the cost-benefit analyses underlying current control programs will need to be revisited. In short, despite increasing need, we have to be even more selective about the programs we undertake.

We will also need to reconsider how alien species are identified. Until now, the introduction of alien species was largely mediated by humans, and their status as non-native was rarely in doubt. In the future, many new arrivals will be species from the northern US moving into Canada on their own in response to climate change (McKenney et al. 2007). This confers considerable ambiguity as to their status.

Under the conceptual framework developed earlier in the chapter, the coming "invasion" of northern US species should be accepted as a natural ecological response to warming temperatures (Walther et al. 2009). These northward-moving species will have an essential role in populating the new climate envelopes that will develop in Canada's southern landscapes, filling niches that our native species are poorly adapted to (Berteaux et al. 2010). Consider the migration of buffalograss. This is a warm-season grass that is widespread in the US Great Plains but rare in Canada (Fig. 9.14). Under warmer and drier conditions, this species will fill new dry grassland niches in Canada that existing species are not well suited to, thereby helping to maintain ecosystem stability and function.

Fig. 9.14. Buffalograss is widespread in the US Great Plains and will move into the southern prairies as temperatures warm, filling new ecological niches. Credit: K. Kenraiz.

Major ecological disruptions are unlikely to arise from northern US migrants (Mueller and Hellmann 2008). These species will be at the northern edge of their range and poorly adapted to cooler Canadian environments (otherwise, they would already be here). Moreover, they will be moving in response to climatic changes rather than invasive tendencies. Nevertheless, exceptions may occur, forcing managers to contemplate control measures. In these cases, it will be the potential for economic and ecological disruption, rather than simply their alien status, that is most relevant (Coops et al. 2008).

Box 9.4. Alien Species as Endangered Species

Under the *Species at Risk Act*, a species must have extended its range into Canada without human intervention and have been present for at least 50 years before it can be recognized under the Act (GOC 2002, Sec. 2). The accommodation of climate change demands a more nuanced approach (Gibson et al. 2009). Species that are at risk in the US should not be ignored for 50 years after they establish in Canada just because they are not historically "ours." On the other hand, it would be inappropriate to list all incoming species as threatened simply because these vanguard populations are inherently unstable. A dynamic and integrated approach is needed.

Species-Level Conservation

Recovery Plans

Species recovery plans have three main components: species information, a set of management objectives, and proposed management actions. Adjustments to each of these components are needed to accommodate climate change.

The species information component should include a formal vulnerability assessment that examines climate exposure as well as sensitivity. This assessment should take both short-term changes (i.e., increased climate variability) and long-term changes (i.e., progressive warming) into account. The potentially positive effects of climate change should also be considered.

Once the preindustrial baseline fades from relevance, a dynamic reference state will need to be used when setting recovery objectives. The procedures for doing so have not yet been worked out, but presumably will entail some form of habitat and population modelling. The concern is mainly with long-term recovery objectives that reference the natural state. This state is now a moving target.

Short-term recovery objectives do not require much adjustment because they focus on population survival and reversing population declines. The relevant reference points, such as minimum viable population size, are determined largely by proximate risk factors.

Finally, recovery plans should include species-specific conservation measures for mitigating climate-related threats and facilitating adaptation.

In the near term, the greatest concern for most species is increased climatic variability and extreme weather events (Parmesan et al. 2000). This issue should be addressed like other conventional threats, through direct mitigation measures. For example, it is expected that an increase in extreme rainfall events will cause more acute food shortages for grassland raptors (Fisher et al. 2015b). This can be mitigated through habitat management that increases the abundance and availability of prey, and when necessary, through supplemental feeding.

The long-range threats posed by climate change are more challenging to address because the amount and timing of the changes are uncertain. There is also less certainty about how effective our actions are likely to be. Finally, it is often difficult to obtain adequate funding because long-term risks may be perceived as less important than immediate threats. In this respect, climate change is similar to other "slow-creep" issues, such as industrial cumulative effects, which have proven difficult to address.

The most effective approach for helping species cope with progressive warming is to increase their resilience and adaptive capacity (Seidl 2014). This entails reducing the current causes of stress, whatever they may be, using conventional conservation measures. It also includes reducing barriers to movement. While not novel, these are essential components of adaptation and should not be overlooked.

Another approach is to slow the rate of ecological transitions to buy time for species to adapt. This runs counter to the concept of facilitating natural transitions, but it may be needed as a temporary measure to help species that are unable to keep pace with rapid warming. For example, we might try to reduce the rate of anthropogenic and natural disturbances (like fire) that mediate ecological transitions. Or we might seek to slow the influx of disruptive species from warmer regions. Forestry replanting programs can help slow transitions in forested landscapes.

Lastly, there are mitigation measures that are part of the conventional conservation toolkit but are used differently under climate change. The two main methods that fall into this category are protected areas and assisted migration.

Protected Areas

The use of protected areas for conserving focal species is far more difficult under climate change because their preferred habitat becomes a moving target (Sandler 2013). Two approaches are available and both have major shortcomings.

The first approach is a "floating" reserve system in which protected areas shift location in synch with changes in the climate envelope of the species of interest. The main drawback of this approach is that it forces us to trade relatively pristine landscapes inside existing reserves for degraded landscapes elsewhere, defeating the core purpose of protection (Rayfield et al. 2008). Furthermore, the logistics of implementing a floating reserve system at broad scales for multiple focal species seem insurmountable.

The second approach is to represent both the current *and* future habitat needs of focal species within a fixed reserve system (Schuetz et al. 2015). This approach is conceptually sound but impractical when the amount of land available for protection is constrained, as it usually is. With even a small number of focal species, a requirement to represent *all* future contingencies could easily entail protecting most of the land base. Another shortcoming of this approach is that it is heavily dependent on long-range projections of bioclimatic envelope models, which are subject to considerable uncertainty, particularly after 2050.

Because of these limitations, focal species representation should be deemphasized in systematic conservation planning initiatives, in favour of the more robust coarse-filter approach. The dilemma for conservation practitioners is that focal species are highly valued by the public and stakeholders and cannot simply be ignored. A solution is to include focal species in the planning process, but mainly for informational purposes. The intent is to determine how well an optimally configured coarse-filter design will protect selected focal species, both now and into the future. If significant deficiencies are noted, then the costs and benefits of alternative designs can be investigated and discussed (preferably in a structured decision-making framework). The high uncertainty of climate envelope projections should be formally included in these deliberations.

Assisted Migration

In contrast to the other conservation measures we have examined, assisted migration (or assisted colonization) entails direct intervention in the adaptation process. It is also one of the more controversial measures and the subject of ongoing debate (Ricciardi and Simberloff 2009; Hewitt et al. 2011). The objective is to help species keep pace with climatic changes by physically moving them to new locations. As such, it can be viewed as an extension of the reintroduction methods we discussed in Chapter 6.

To date, assisted migration has been applied mainly in the forestry sector, where the focus is on maintaining forest productivity. BC is leading the way in Canada, with its development of a new climate-based seed transfer system (O'Neill et al. 2017). Seed transfer systems provide the rules forestry companies must follow when sourcing seed stock for their reforestation programs. These rules ensure that the seed stock is matched to the local climate, which in the past has meant limiting seed transfers to their zone of origin. To facilitate adaptation to climate change, BC's new system allows companies to use stock from zones that are slightly warmer. Other provinces are contemplating similar changes but are at an earlier stage of implementation (Klenk 2015).

Because BC's new seed transfer system involves moving populations within the existing range of a species, there is little concern about invasiveness. Moreover, rather than targeting some distant point in the future, the system is designed to match seed stock to the climate conditions 12–17 years from the present (O'Neill et al. 2017). The intent is mainly to help the selected species "catch up" with the climate shifts that have already occurred. As a result, the implementation of this system has not been very controversial (Klenk 2015).

It is possible that more aggressive assisted-migration programs will be implemented by the forestry sector in the future. The field is evolving rapidly, and considerable research is underway, including growth trials of non-native species (Thorpe et al. 2006). The inherent risks have yet to be fully assessed, and it is possible that implementation efforts will at some point face public resistance.

Assisted migration is also being explored as a conservation measure for maintaining biodiversity (Gallagher et al. 2015; Vitt et al. 2016). There are several situations where assisted migration might be used. One involves species that are unable to keep pace with moderate to fast rates of climate change because their capacity for migration is low. If conditions along the trailing edge of their range become clearly unsuitable, leading to population declines, it would be better to move the affected populations to a more viable area rather than simply allowing them to become extirpated.

Another situation involves species that are contracting to the core of their range because of anthropogenic stress (recall the burrowing owl example). These species are unlikely to keep pace with changes in their climate envelope without human intervention.

Finally, assisted migration may be warranted for species that are unable to shift their range effectively because of an anthropogenic barrier. Species at risk in southern Canada are a prime example. Many of these species may require assistance in shifting their range northward because of the barrier presented by agricultural lands.

The use of assisted migration for biodiversity applications is still at an early stage, and there are a variety of challenges to be overcome (Hancock and Gallagher 2014; Gallagher et al. 2015). The issue that has received the most attention, and has been the main cause of controversy, is the potential for the transplanted species to disrupt the recipient ecosystem. Species transfers have a checkered history, often involving unintended consequences. For example, the red squirrel was introduced to Newfoundland in 1963 in an attempt to bolster the island's struggling American marten population (Benkman 1993). Unfortunately, the introduction of the squirrels, which are not native to Newfoundland, led to increased competition for conifer cones which hastened the decline of Newfoundland's threatened red crossbill (COSEWIC 2016).

Proponents of assisted migration maintain that climate adaptation applications are not comparable to past species introductions and carry much less risk (Schlaepfer et al. 2009; Vitt et al. 2009). Assisted migration is expensive and time-consuming, so it is unlikely to be used for species that can shift their range without assistance. This eliminates species with invasive potential. Furthermore, the target habitats lie along climate gradients that the species could and would reach naturally if not for anthropogenic barriers and ecological stressors. Finally, the intent is not to place species into new ecosystems, but to help them maintain their position in existing ecosystems as they shift location in response to climate change.

The other factors to be considered are common to all species reintroduction programs (see Chapter 6). These include knowledge about the ecology of the species, knowledge about effective translocation methods, the availability and viability of founder populations, the availability of suitable transplant locations, current and future threats to population viability, and stakeholder and institutional support (Gallagher et al. 2015). Cost is an overriding factor and will likely limit the widespread application of assisted migration, just as it limits reintroduction and restoration projects today.

In summary, both the promise and risks of assisted migration are less than they might appear. Given capacity constraints and ecological limitations, the reality is that most species in most areas will have to adapt to climate change on their own. Nevertheless, assisted migration is a conservation tool that is worth exploring as part of the strategic planning process for species at risk. As always, the question for managers is how the costs and benefits compare to other available management options (Schlaepfer et al. 2009). Pilot studies are a logical first step, providing the information needed to make informed decisions about broader applications (Fig. 9.15).

Fig. 9.15. The northern blazing star is a candidate for assisted migration because it faces barriers to migration and has high habitat specificity. Transplantation trials are underway in northern Alberta, in sites that are several hundred kilometres north of the species' current range (Wang et al. 2019). The objective of these trials is to determine the feasibility of assisted migration and to investigate the factors influencing translocation success. Credit: S. Nielsen.

Population-Level Triage

In Chapter 6, we discussed the concept of conservation triage in the context of prioritizing species. Triage has also been proposed at the population level, as a response to climate change (Lawler 2009; Hagerman and Satterfield 2014). The basic argument is that, for many species, populations along the trailing edge of the range are unlikely to persist under warmer temperatures. Therefore, to obtain the greatest benefit from limited conservation resources, we should redirect conservation resources from trailing-edge populations to leading-edge populations.

The conservation of salmon along the Pacific coast of Canada and the US provides a case in point. Hundreds of millions of dollars are spent annually on the conservation of salmon up and down the coast (Barnas et al. 2015). Because of climate change, the maximum summer water temperature in many southern streams is now approaching and occasionally exceeding the thermal limits of salmonids (Honea et al. 2016). As temperatures continue to increase over the course of this century, southern salmon populations are expected to decline, despite restoration efforts (Fig. 9.9). Coincidently, various salmon species are now being observed in Arctic waters, indicating that losses of southern populations are likely to be offset by gains in northern areas (Nielsen et al. 2013). From this broader perspective, a triage approach that redirects conservation resources from southern to northern populations seems warranted.

While these arguments are compelling, a major shortcoming is that they frame the decision too narrowly. As we saw in Chapter 6, triage solutions will not be broadly optimal if important dimensions of the problem are omitted.

In this case, attention is narrowly focused on the local persistence of individual populations while other relevant factors, such as genetic adaptations and range shifts, are not considered.

If we abandon warm-adapted trailing-edge populations, allowing them to become quickly extirpated, we stand to lose part of the gene pool (Savolainen et al. 2007; Martins et al. 2012). As a result, the species may experience a permanent contraction in range because it can no longer accommodate the same spectrum of environmental conditions (Fig. 9.16; Hampe and Petit 2005). Thus, trailing-edge populations still merit conservation, even though they may not persist in their current locations (Hampe and Petit 2005; Battin et al. 2007). The objective is to maintain these populations long enough for gene flow to occur into northern regions.

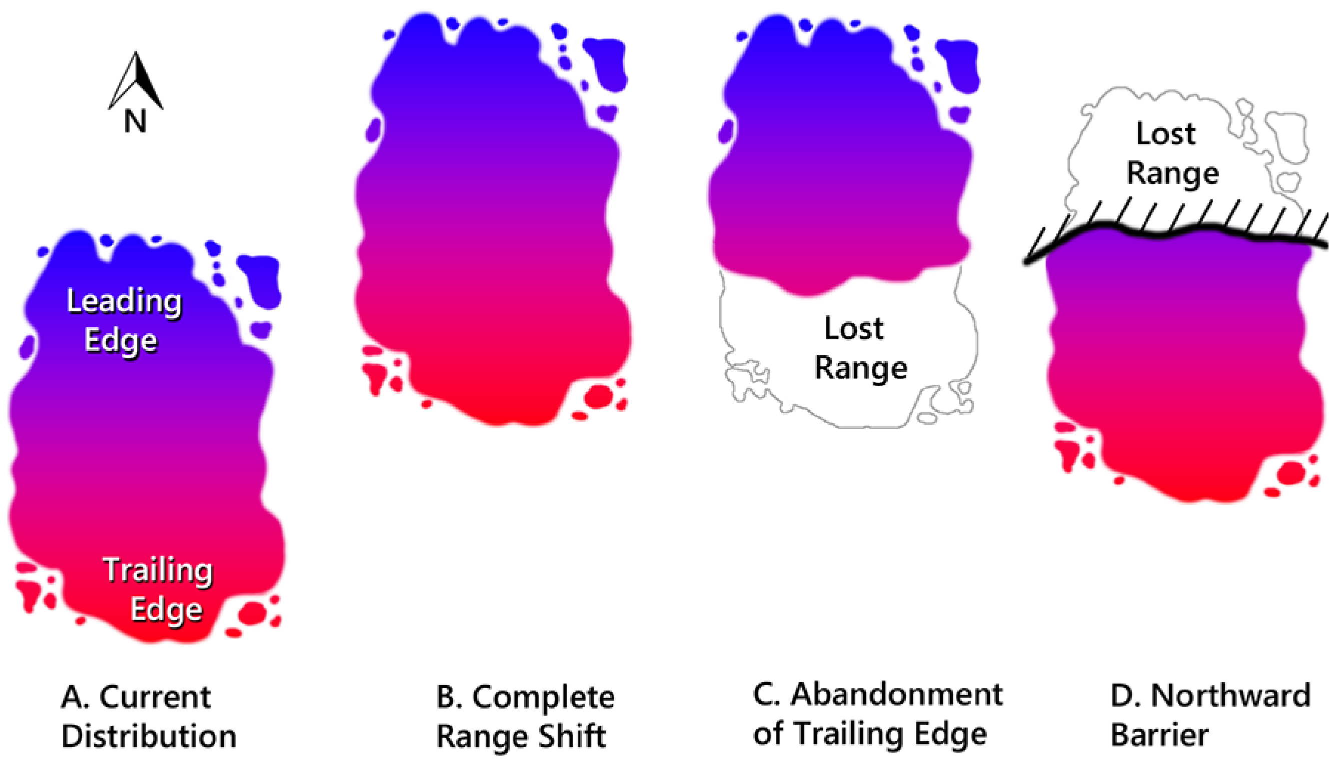


Fig. 9.16. Many species exhibit genetic adaptations to local climatic conditions, illustrated here in a red (warm) to blue (cool) colour spectrum. Panel A represents the current distribution of a hypothetical species. Panel B represents its future distribution under a warmer climate, assuming that the range shift is not impeded and all genetic variation is maintained. Panel C illustrates the future distribution that may arise if trailing-edge populations are allowed to become extirpated. The range has contracted because the gene pool is smaller and cannot accommodate the same spectrum of environmental conditions. Panel D illustrates an alternate scenario in which a barrier prevents the species from shifting its range northward. The species remains viable because warm-adapted populations replace cool-adapted populations, but the range is compressed.

Another consideration is the practicality of the funding reallocations that are central to the triage approach. Even if population-level triage provides a clear conservation benefit, this does not mean local authorities will necessarily agree to transfer funding to distant jurisdictions over which they have no control or responsibility. The political optics of formally abandoning a local population are also highly problematic for regional governments.

These shortcomings do not mean triage is without merit. As conservation practitioners, we should always seek to optimize the allocation of limited conservation resources (Bottrill et al. 2008). And we must accept that this will sometimes involve unpalatable choices. However, it is important to consider all relevant dimensions of a management problem. Furthermore, the progressive nature of climate change demands a dynamic framework for management—the transitions are just as important as the endpoints. Finally, the practical reality is that optimal resource allocation will always be easier to achieve at the local level, where organizations are allocating resources within their own domain. For issues that cross jurisdictions, we usually have to settle for coordinated planning rather than fully integrated planning.

CHAPTER X

STRUCTURED DECISION MAKING

Structured Decision Making

In the last few chapters, we have examined the practical methods used for maintaining biodiversity. But there is more to conservation than linking threats with mitigation measures. Conservation is fundamentally a social enterprise. In most applications, trade-offs between conservation objectives and other social objectives must be addressed before any action can be taken. Also, in the face of constraints, it is necessary to determine which conservation objectives are most important and which conservation methods are best suited for a given application. In short, applied conservation entails making decisions.

In this chapter, we will work through the decision-making process, step by step. As our guide, we will use the framework referred to as structured decision making (SDM). This approach is widely accepted as the best practice standard for making management decisions involving multiple objectives (Runge 2011; Gregory et al. 2012; Conroy and Peterson 2013). Many variants of SDM exist, tailored to specific applications (e.g., CMP 2013; Carwardine et al. 2019). We will remain focused on the general principles, which have broad applicability.

General Principles

SDM draws on insights from decision science and cognitive psychology and has been tested and refined through years of application. It is a highly practical approach, described by Gregory et al. (2012) as follows:

> SDM is a useful way to deal with the realities of everyday environmental management … SDM reframes management challenges as choices; not science projects, not economic valuation exercises, not consultation processes or relationship builders. You have a decision (or a sequence of decisions) to make. The con-

> text is fuzzy. The science is uncertain. Stakeholders are emotional and values are entrenched. Yet you—or someone you are advising—has to make a choice. (p. 2)

In conservation applications, choices come in three main forms, all of which are amenable to SDM:

1. **Selection:** deciding which action to take when multiple alternatives are available
2. **Prioritization:** ranking options (e.g., for resource allocation)
3. **Identification of breakpoints:** determining when to act or how much of a certain action is needed

A key feature of SDM is that it integrates the social and scientific dimensions of conservation decision making. Through SDM, we identify solutions that optimally address multiple objectives while simultaneously building support for implementation. SDM's effectiveness arises from the structure it provides for thinking through complex problems and navigating difficult trade-offs. The emphasis is on clarifying objectives and understanding the full consequences of management alternatives. The resulting decisions are rigorous, inclusive, defensible, and transparent. The SDM process also reveals key uncertainties and knowledge gaps, providing guidance for research efforts.

Fig. 10.1 presents an overview of the SDM framework. The process begins with decision framing, which establishes the context and scope of the decision. This is followed by a sequence of structuring steps—the heart of SDM—leading to the selection and implementation of a preferred approach.

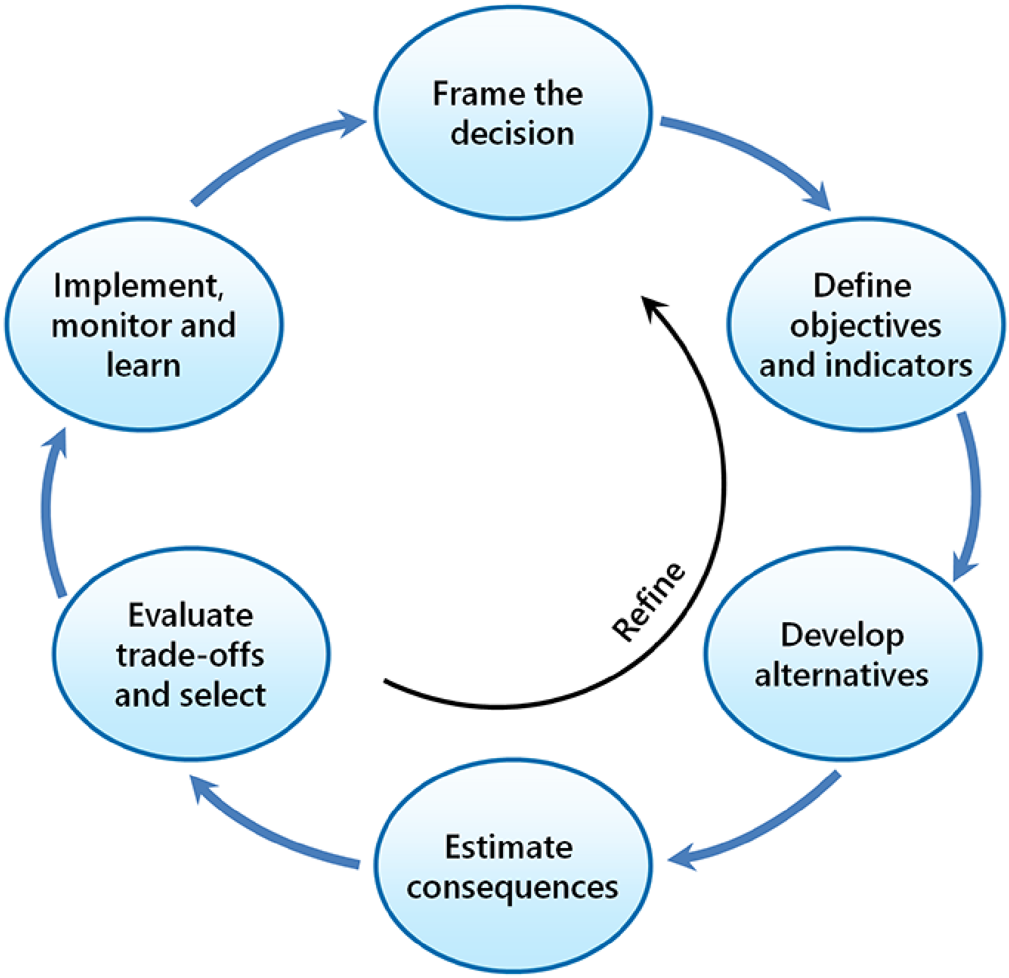


Fig. 10.1. The SDM framework, adapted from Gregory et al. 2012.

A useful tool for organizing and visualizing the decision process is a consequence table (Gregory et al. 2012). As illustrated in Table 10.1, a consequence table consists of a set of objectives (rows) and a set of potential management approaches that can be used to achieve those objectives (columns). The cells of the table are filled with predictions of how the management alternatives are expected to perform against each of the objectives. We will work through a simple example as an introduction to the SDM process. This example will serve as a roadmap for the details that follow in the rest of the chapter.

Table 10.1. A simplified SDM consequence table for caribou management.

Objective	Approach A	Approach B	Approach C
Caribou viability	Outcome A1	Outcome B1	Outcome C1
Timber harvest	Outcome A2	Outcome B2	Outcome C2
Indigenous hunting	Outcome A3	Outcome B3	Outcome C3

For our example, will use the development of a management plan for woodland caribou. The context is provided by the federal woodland caribou recovery strategy, which sets forth several recovery objectives. Our task is to determine which management action, or combination of actions, should be applied to achieve these objectives in our management area. To be clear, this example is only meant to illustrate how the SDM process works. All of the complexities have been stripped away. In Case Study 3, we will examine a real-world example of caribou recovery planning.

The first step is to define the management objectives, which describe the desired outcomes of a given planning process. Objectives express what matters to us. Under SDM, objectives are defined not only for conservation-related values, but also other social values affected by the decision. In our example, caribou viability is identified as a core objective, as per the recovery strategy (Table 10.1). But objectives related to forestry and Indigenous hunting have also been included because these activities might be affected by recovery measures.

When defining the objectives, we must differentiate the ends (the outcomes we seek) from the means (methods for achieving the ends). We must also specify how the objectives will be measured, allowing the performance of proposed management approaches to be compared. In our example, we might use caribou population size or growth rate as an indicator of caribou viability.

The next step is to develop alternative management actions or strategies that could be used to achieve the objectives. Depending on the situation, these might be distinct management actions, different combinations of actions, or alternative levels of the same action. For example, in Table 10.1, Approach A might represent current management, Approach B might involve innovative industrial practices, and Approach C might involve establishing new protected areas. Hybrid approaches are also possible.

The preferred approach is identified by predicting outcomes for every alternative and then determining which offers the best overall result. The prediction component is meant to be objective and is heavily dependent on science. In contrast, the selection of the best option is generally a matter of social choice because value-based trade-offs are usually involved. In our example, we might use a statistical model to predict caribou population trends under each management alternative. Timber flow and Indigenous hunting opportunities would also be predicted. Decision makers, with input from stakeholders, would then evaluate the overall performance of each approach and make a choice.

Though the SDM process is described here as a linear process, in practice it usually involves iterative refinement. Insights from the prediction and assessment stage may lead to changes in the set of management alternatives and possibly new objectives.

The SDM process is designed to reveal win-win solutions if they exist. However, in most conservation applications, compromises are necessary (McShane et al. 2011). The trade-off between caribou recovery and forest harvesting is a good example. In such cases, decision makers must determine the appropriate balance among objectives—a political decision that typically disappoints some or all stakeholders. SDM is still of value in these situations because it clarifies the values at stake and the likely outcomes of management alternatives. It also brings fundamental trade-offs into high relief. Thus, even if the decision must be made politically, it will be properly informed.

The last stage in the process is implementing the plan and monitoring the outcomes. Monitoring provides insight into the validity of planning assumptions and predictions. It can also be used to support active learning about the system and how it responds to manipulation. The information gained is used to inform future iterations of the planning cycle.

Box 10.1. Decision-Making Terminology

Decision. In SDM, a decision is a judgment or choice involving multiple objectives and alternative courses of action.

Goals and objectives. Goals and objectives describe desired outcomes. Goals are generally broad statements of what is desired. Objectives are subcomponents of a goal and are specific and measurable.

Plan. A plan is an organized collection of decisions supporting a common purpose. It is a roadmap to a desired future, describing where we are going (objectives) and how we will get there (methods).

Strategy. A strategy is similar to a plan; however, strategies have a broader scope and are more abstract than plans. Plans tend to be more operationally focused.

Trade-off. A trade-off arises when a constraint or incompatibility prevents the simultaneous achievement of multiple objectives. One must give up one thing to achieve something else.

Values and preferences. Values are what we fundamentally care about and believe to be important. Values tend to be relatively fixed. Preferences are defined in terms of the trade-offs we are willing to make in a particular situation. They change with the circumstances.

The Decision Hierarchy

The full process of SDM, from problem identification through to implementation, is largely synonymous with the concept of planning. We can think of a plan as a container for one or more SDM decisions supporting a common purpose.

In resource management, planning is hierarchical, and different terms are used to identify the different levels (Fig. 10.2). Policies are at the top of the hierarchy. They establish broad goals and priorities and provide high-level guidance for action. At the bottom are operational plans, which are implementation oriented. In between are layers of strategic plans that translate the broad direction provided by policy into a region or issue-specific context. For example, a polar bear management strategy represents an intermediate step between a policy commitment to maintain biodiversity and the setting of annual polar bear hunting quotas.

Fig. 10.2. A diagrammatic representation of the decision-making hierarchy for resource management.

Policies, strategies, and operational plans exist along a continuum that lacks clear demarcations. Moving along this continuum, from policy to operations, the scope narrows and the level of detail increases. In addition, decisions become more technical, with less need for social deliberation, since we are increasingly implementing choices made at higher levels.

Although policies are intended to provide context and direction for lower-level initiatives, these linkages are not always clear. As we saw in Chapter 7, the institutional mechanisms for integration are often deficient. Therefore, it frequently requires considerable effort on the part of planners to interpret the broader context they are working within (which is part of decision framing).

Most types of decisions can benefit from SDM; however, high-level government policy decisions are often too broad and complex to structure effectively. High-level policy is usually created through a political process, as described in Chapter 3. SDM also has limited applicability for routine operational decisions that are essentially algorithmic (e.g., the determination of an annual harvest sequence, as directed by a broader forest management plan).

It should be noted that SDM is not limited to government decision making. It can help any organization faced with making decisions that involve difficult trade-offs. For example, a conservation group facing capacity constraints might use SDM to choose among potential projects. Furthermore, the level of detail and effort involved in implementing SDM can be scaled up or down to match the scope and importance of individual applications.

The Role of Conservation Practitioners

Conservation practitioners participate in decision-making processes in three distinct roles: stakeholders, advisors, and decision makers. Practitioners are considered stakeholders when they engage in advocacy, promoting conservation as a planning objective. Such input falls within the social domain of decision making. Practitioners are considered advisors when they contribute technical expertise about how conservation objectives can be achieved. This input falls within the scientific domain of decision making.

The reason for drawing a distinction between advocacy and technical advice is that the social and scientific aspects of decision making need to be handled differently, as we will see in the sections that follow. In principle, there is no reason why a conservation practitioner cannot contribute to both aspects of a decision, so long as the necessary distinctions are maintained. However, decision makers and stakeholders may consider technical advice biased and unreliable if it is coming from an advocate (see Chapter 4).

Conservation practitioners also serve as decision makers, mainly for conservation-oriented planning initiatives in the middle to lower levels of the decision hierarchy. Such initiatives usually take place under the auspices of an agency with a mandate for implementing conservation policy, typically within government, but not exclusively so. Some initiatives may represent the implementation phase of a higher-level strategy, whereas others may be responses to local conservation problems. The case studies in Chapter 11 provide examples of decision making by conservation practitioners in a variety of settings.

Decision Framing

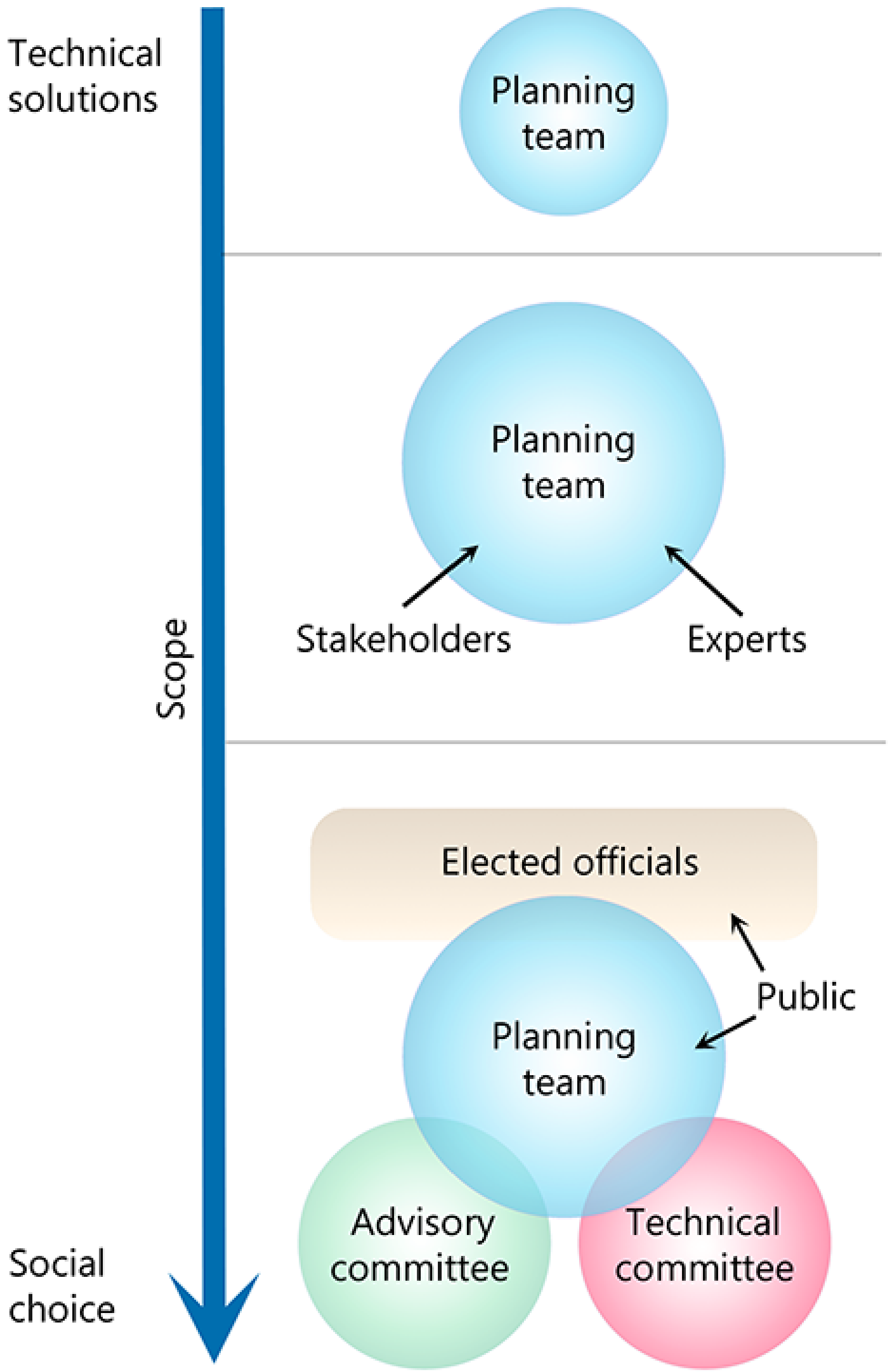
Structured decision making begins by establishing a **decision frame** that defines what the decision is about and sets bounds on what will be considered. The main elements of the decision frame are (1) the context and purpose of the plan, (2) its scope, and (3) the process by which decisions are made.

A plan's context and purpose constitute its foundational elements. This is where the motivation for the plan and its role in the broader decision-making hierarchy are defined. In most cases, there is some form of problem to be resolved or a desired outcome to be achieved. This is often predetermined by a higher authority.

A plan's scope describes the range of objectives and management options under consideration. It also defines the planning area and time horizon. The intent is to establish boundaries to keep the planning process focused and manageable. There is a limit to how much complexity can be accommodated before the process bogs down. High-level plans with a broad scope must generally sacrifice detail to remain tractable. Refinement of the scope may occur in later stages of planning, as the problem becomes better understood; however, the process should not be allowed to drift.

The scope should reflect the planning agency's mandate, its capacity for planning (including time, funding, and expertise), and the extent of its authority. Other constraints, including existing laws and policies, also need to be respected. If the planning agency cannot ensure that the management options under consideration can be implemented, then either the planning agency or the scope should be changed. Regional planning presents the greatest challenge because, as we saw in Chapter 7, management authority at the regional scale is fragmented among sectors and jurisdictions.

The decision frame also describes how decisions will be made. The main questions are: Who will be involved? How will they be involved? And, how will control over decision making be exercised? Operational planning can often be handled internally, within the planning team. But as the scope of planning broadens, there is an increased need for social input and interdisciplinary technical expertise (Fig. 10.3). External voices bring new facts, ideas, and perspectives to the table, resulting in more informed decisions and a greater likelihood of successful implementation (Gray et al. 2012).



1. Small planning teams rely primarily on internal expertise. This approach is common for operational planning within a narrow well-defined scope (e.g., the restoration of a selected wetland site).

2. Large planning teams may conduct planning internally but often obtain advice from external experts and periodically solicit feedback from stakeholders. An example is the development of a species recovery strategy.

3. Complex planning systems may include a core planning team, a stakeholder advisory committee, and one or more technical committees that operate in concert. Insight into public opinion may be obtained through outreach efforts. Recommendations from the planning team are passed on to elected officials who make the final decision. This type of planning is common with high-level decision making, such as regional planning.

Fig. 10.3. Planning systems exist along a gradient of increasing scope and complexity. This diagram illustrates three common forms, arranged in order of complexity. Many variants of these systems exist in practice.

Obtaining Social Input

There are two main approaches for obtaining social input. One is to collect input through surveys and requests for public feedback on draft plans. The benefits of this approach are that it provides broad insight into public values and preferences and it is relatively straightforward to administer. The main shortcoming is that the responses tend to be superficial and are contingent on context. The answers depend on how the questions are framed and how much the respondents know about the issue.

Another option for obtaining social input is to work directly with stakeholders. Stakeholders present distinct perspectives and are generally motivated and knowledgeable about the issues. In addition, they often have a central role in implementation and can provide insight into the pros and cons of management alternatives and their consequences.

Integrating stakeholders into the planning process allows for structured dialog among competing interests. This can be an effective way of finding broadly acceptable solutions to social trade-offs. However, it can be challenging to ensure that the full spectrum of societal values and priorities are represented, especially values related to biodiversity. Stakeholder groups provide depth at the expense of breadth. Incorporating stakeholders also makes the planning process more complex, time-consuming, and costly (Gray et al. 2012).

It is, of course, possible to do both: solicit broad public input and work with stakeholder committees. As a rule, the amount of input needed is a function of the scope of planning (Fig. 10.3). In practice, cost and capacity constraints limit what can be accomplished.

Two forms of stakeholder engagement are most common. In the first, a stakeholder advisory committee provides ongoing advice to the planning team but is not involved in the actual crafting of the plan. In the other approach, stake-holders are integrated into the planning team, and the plan that is developed is forwarded as a set of recommendations to a higher level of authority for final decision making.

Indigenous people merit special mention because they cannot be treated as typical stakeholders. When planning involves traditional lands, consultation with affected Indigenous communities is legally required, and certain standards must be met (SCC 2017a). In northern areas, land claim agreements provide additional rights (Wyatt et al. 2013). Some areas now feature co-management programs.

Good facilitation and clear expectations are critical to productive stakeholder engagement. The dialog must be kept on track and focused while still ensuring that all concerns are addressed. The aim is to build a common understanding of value sets and priorities through group learning, and then to channel this into effective problem solving. Case Study 5 provides a useful example.

In most cases, the government will retain control over the final decision because it is ultimately accountable for decisions concerning public lands. Moreover, there is a need to balance the interests of stakeholders, which tend to be locally focused, with broad societal values and priorities. In government-led planning, the line of accountability runs from the responsible minister down through line departments and on to individual planning teams. The specific level where the final decision will be made should be identified as part of the decision-framing process.

Objectives and Indicators

Setting Objectives

SDM is an outcome-oriented approach, meaning that objectives, rather than threats or problems, are the central focus. We define what we want, then identify the best way to achieve it, dealing with conflicts and barriers as they arise. For this approach to be effective, the set of objectives must be clearly specified and comprehensive (Failing and Gregory 2003; Game et al. 2013). Objectives should address the core purpose of the planning initiative as well as other values that may be affected by management actions. The aim of being inclusive is to permit potential conflicts to be dealt with within the SDM process. Not doing so risks failure at the time of implementation (which has been the fate of many conservation initiatives in the past).

It is not always possible to specify the complete list of objectives at the outset. Values that are secondarily impacted may not be known until later in the process, when management options have been fully specified. SDM handles this issue using an iterative approach. The process begins with an exploratory sketch that maps out the decision structure at a coarse level. The preliminary list of objectives is then refined as detail is added to the decision structure in successive iterations.

When selecting objectives, the desire for inclusiveness must be balanced against practicality. There are limits to how much detail can be accommodated. Secondary objectives should be weeded out if their presumed sensitivity to management actions is not validated at the assessment stage. Also, while some objectives may have subcomponents that need to be distinguished, there should be no double counting. The test is to ask if anything important would be overlooked if we were to substitute one sub-objective for another.

We must also take care to distinguish between ends and means. Ends are what we fundamentally want—our true objectives. Means are methods for achieving the ends, and they are captured in the SDM process as management alternatives. In practice, making this distinction can be challenging because ends and means exist in a hierarchy—the means described in one plan may serve as the ends of a lower-level plan. Much depends on how the decision is framed.

Another consideration is the level of detail used to describe the objectives. On the one hand, we want enough detail to ensure that the desired outcomes are clearly understood and not subject to alternative interpretations. On the other hand, getting too specific, through the use of detailed quantitative targets, can be counterproductive because it reduces decision-making flexibility (Martin et al. 2009). The aim is to express what we want without creating all-or-nothing scenarios. The possibility of compromise solutions needs to be left open, as more often than not, these will be the only workable solutions.

The relative weighting of objectives has been purposely omitted from our discussion so far. Although objectives do vary in their importance, formal weighting should not occur until the decision stage. Throughout the rest of the process, objectives should remain on a level playing field so that the search for optimal solutions is thorough and robust.

When it comes to specifying conservation objectives, the broad goal of maintaining biodiversity needs to be translated into ecologically meaningful working objectives relevant to the decision scope. In doing so, it is important to not lose sight of the conservation "big picture." The individual outcomes sought within a given initiative should meaningfully contribute to the whole. If it is unclear how they do so, there is a problem. Trade-offs among conservation objectives and the effects of climate change present special challenges that must not be overlooked. These tasks require the technical expertise of conservation practitioners.

Conservation practitioners should also ensure that conservation objectives accurately reflect a "nature-first" perspective, uncontaminated by pre-emptive compromise (Tear et al. 2005). Even though such objectives will not always be achievable, they represent the appropriate starting point for deliberations. Clarity about what we *really* want enables decision makers to understand and reflect on the true cost of any compromises that may be required. This guards against the **shifting baseline** scenario, or "ratchet" effect, where the acceptance of a degraded state as a management norm leads to progressive declines over time (Pauly 1995).

Selecting Indicators

Once objectives have been defined, suitable indicators for measuring them need to be identified so that the relative performance of the management alternatives can be assessed. Indicators are also used in monitoring programs and research projects (discussed below).

There are several basic characteristics that all good indicators share, regardless of the application (Duinker 2001; Tear et al. 2005). Good indicators are:

- **Clear.** The meaning of what is being measured is broadly understandable and not subject to alternative interpretations
- **Reliable.** The indicator accurately measures what it is intended to measure (i.e., free from bias) and repeated measurements generate the same results
- **Practical.** Cost and technical feasibility are not significant barriers
- **Relevant.** The indicator is sensitive to the processes of interest and can help discern important differences among management alternatives

An additional concern when selecting indicators is comprehensiveness. Like the proverbial blind men examining an elephant, we may be misled if we only see part of the full picture. It may take multiple distinct indicators to obtain a complete characterization of the entity we are measuring.

Developing a complete characterization can be challenging when working with amorphous management objectives (as many are). How are we to measure something comprehensively when its meaning is open to interpretation? The answer is that the selection of indicators is part of the interpretive process. The choices we make serve to crystallize the meaning of the objectives being measured within a given decision-making context. Thus, indicators are not only used to measure objectives, they also help us define objectives in practical terms.

Consider the selection of indicators for caribou recovery. Is it sufficient to view this objective through the lens of population size? Perhaps long-term viability is important as well. What about spatial distribution? Is it ok if all our caribou are in a zoo? If not, then perhaps we need some measure of distribution relative to the historical range. But then how do we account for climate change?

What this line of questioning illustrates is that indicator selection is actually an extension of the objective-setting process. It includes both technical and subjective elements that together give concrete, practical meaning to our objectives.

The desire for comprehensiveness must be balanced against tractability. A long list of indicators may help to characterize an objective, but it seriously complicates the task of assessing management alternatives, particularly when multiple objectives are involved. There is a limit to how much information can be accommodated before overload occurs. Thus, restraint is necessary; we should "count what counts," and not more. Each indicator should represent a distinct and important aspect of the objective, without redundancy.

Lastly, indicators are used in decision-making applications to predict the *future* state of objectives under alternative management approaches. Predicted performance provides the basis for deciding which approach is best. This adds another dimension to the selection process. Not only should indicators be technically sound and complete, their response to management actions should be understood so that meaningful predictions can be made.

As might be expected, it can be difficult to find indicators that meet all of the above criteria. It is often necessary to make do with indirect measures. And for some objectives, especially those associated with cultural or esthetic values, quantitative measurement may not be possible. For example, it may be difficult to obtain a meaningful numerical measure of the quality of recreational experiences. In these cases, a qualitative indicator using a constructed scale can be used (e.g., very good, good, average, bad, very bad). As a rule, it is better to make the best of a rough estimate than to omit an objective from consideration. This ensures that the decision-making process is inclusive and robust. For such indicators, it is important to include clear descriptions of each level so that there is a common understanding of their meaning.

Conservation practitioners support this stage of the decision-making process by providing the ecological expertise needed for selecting appropriate indicators for biodiversity objectives. This includes knowing what options are available, in the form of direct and indirect measures. It also includes the ability to judge which indicators will be most effective in terms of reliability, practicality, and accurately representing the desired biodiversity outcomes.

In higher-level planning initiatives, there may be considerable pressure to simplify the conservation dimension of the decision. It falls to conservation practitioners to ensure that the central concerns of conservation are not lost in the process. Proposals to capture biodiversity objectives using broad indicators of environmental health or ecosystem services are inappropriate. These measures are not proxies for biodiversity (see Chapter 4). Nor is it acceptable to focus exclusively on a single species because it has a high public profile or is the centre of a legal dispute. Finally, biodiversity indicators should not be based on species richness or simplistic summary measures of overall biodiversity status (Devictor and Robert 2009). Such composite measures tend to mask the elements of biodiversity that are of primary conservation concern (see Box 7.2, **Chapter 7**). Instead, indicators should focus on the components of biodiversity that are sensitive to human disturbance and require protection.

Developing Management Alternatives

Management alternatives represent different ways of achieving the set of objectives, each with its own strengths and weaknesses. Having a wide range of options to choose from increases the chances of finding an approach that is broadly suitable.

The first step in developing alternatives is to identify or devise actions well suited to achieving individual objectives, without regard to trade-offs with other objectives. Doing this fosters creativity and breaks us free from preconceived solutions. To keep the process manageable, the boundaries established by the decision frame should be respected. It is also appropriate to screen out activities that are clearly unworkable because of legal constraints, technical infeasibility, or prohibitive cost.

Screening should be done cautiously. It is not always obvious what might work and what might not. Sometimes, finding an effective approach entails pushing back or working around existing rules and other constraints. On the other hand, wasting time exploring dead ends is inefficient. Moreover, there are practical limits to how many management alternatives can be explored within a given planning initiative. Ultimately, a balance must be achieved.

To enable comparisons, the alternatives must have the same spatial and temporal scope, and they must incorporate all objectives. For example, if we want to explore wolf control as a caribou management technique, we still need to specify how forestry will be conducted. Failing to do so would leave an empty cell in the consequence table, disrupting our ability to make comparisons.

Some planning initiatives, especially those related to resource development, may involve activities that unfold over time. In these cases, management alternatives take the form of scenarios that describe the trajectory of development (Francis and Hamm 2011).

The selected management alternatives should be logical, practical, and structured to expose fundamental trade-offs. A useful approach is to develop alternatives around themes, some of which emphasize certain objectives over others, and others that represent creative, balanced approaches. It is also useful to have an alternative that is based on current practices, to provide a common point of reference.

For objectives related to conservation, an effort should be made to identify preventative measures that support resiliency and address root causes, rather than focusing solely on reactive measures and immediate threats (Seidl 2014).

Once preliminary alternatives have been developed, they normally undergo iterative refinement based on insights from the assessment stage. This refinement process presents an opportunity for creative thought. It may be possible, for example, to find workable fixes for weaknesses that are identified in otherwise promising alternatives. Or, the best aspects from different alternatives might be combined into a hybrid approach. At the same time, alternatives that are demonstrably inferior to other options can be weeded out. The aim is to end up with three to five promising alternatives that can be advanced to the final decision-making stage.

Predicting Outcomes

The next step in decision structuring is to predict potential outcomes under each alternative, filling in the cells of the consequence table (Table 10.1). By design, social values have no direct role in this part of the decision-making process. The emphasis is on making robust predictions to provide the basis for selecting a preferred management option. This is what is meant by evidence-based decision making.

The task of predicting outcomes usually requires specialized knowledge. In some cases, the required expertise may be held by members of the planning team. In other cases, planners may solicit input from relevant experts on an ad hoc basis. For complex decisions, a dedicated technical team may be established to work alongside the planning team.

The foremost concern when making predictions is reliability. Good decisions require good information. In the past, a science-based approach was seen as necessary and sufficient. In recent years, there has been a trend toward also incorporating local knowledge held by individuals living or working in the planning area (Failing et al. 2007). This includes the traditional ecological knowledge of Indigenous people as well as knowledge held by resource companies, local residents, hunters, anglers, trappers, and so forth.

The rationale for incorporating local knowledge is that it can fill gaps in scientific knowledge, leading to better predictions (Failing et al. 2007). There is also a social dimension. Stakeholders are unlikely to support a decision if their views about potential management outcomes are dismissed out of hand.

The challenge for decision makers is that accepting all sources of information uncritically is inconsistent with the aim of generating predictions that are as reliable as possible. The solution is to be inclusive when assembling information and then select the best of what is available using pre-established criteria concerning reliability and utility. The selection criteria can be expressed as a series of questions:

- For observations, what steps have been taken to minimize observer bias, measurement error, and the role of chance?
- For conclusions, are the inferences supported by factual data? What steps have been taken to minimize bias and errors in logic?
- Is the information quantitative or qualitative? How much detail is provided?
- How appropriate is the spatial and temporal scope of the information relative to the proposed application?
- What type of vetting process has the information been subjected to?
- Has the information been organized and summarized or is it in raw form?

In controversial cases, competing sources of information can be treated as alternate hypotheses and explored in tandem. This can be considered a form of sensitivity testing, which we will discuss below.

Modelling Approaches

Humans are adept at making predictions about future events on the basis of past observations and accumulated knowledge, but there are limits to how much information we can mentally store and process. Moreover, our inferences suffer from several well-known mental biases, such as overweighting recent events (Kahneman 2011). A variety of decision support tools have been developed to overcome these limitations and they should be used when time and capacity permit (Addison et al. 2013). We will refer to these tools under the collective heading of modelling approaches.

Two types of models are commonly used to support resource management decisions. First, and most common, are statistical models that mathematically summarize important relationships in the data obtained from observational studies (see Chapter 6). An important benefit of these models is that the reliability of their predictions can be expressed in the form of confidence limits. Their main limitation is that they are inflexible. Statistical models cannot be used to model dynamic processes, nor can they be extended to include new variables.

The second type of model is a **process**, or "**stock and flow**," model. These models track the evolution of selected system components under the influence of one or more driving variables (Box 10.2). The internal mechanisms of process models are usually based on findings from observational studies. In many cases, locally obtained information is lacking, so information from other locations must be used, which adds uncertainty to the model.

When a modelling approach is used, an ongoing dialog between modellers and planners should be established. In the early stages, modellers can help planners understand the system they are working in, and its limitations. This can facilitate the refinement of objectives and management alternatives. In later stages, modellers provide predictions about the performance of management alternatives and convey the uncertainty associated with these predictions.

Information also needs to flow from planners to modellers (Addison et al. 2013). Decision support models are not the same as research models. The intent is to provide decision makers with the information they need to assess and compare the available management options. Modellers should be responsive to the needs and priorities of planners and incorporate local information when appropriate. Furthermore, the model cannot be overly complex or opaque. A "black box" model that must be taken on faith is unlikely to be perceived as legitimate by planners and stakeholders.

When modelling results are presented, a reference point should be included to illustrate their significance. For example, the opportunity costs of protection might be described as a percentage of regional resource revenue instead of just a dollar value. For conservation objectives, it is useful to present results relative to an ecological benchmark or a legal requirement. As a general rule, multiple framings should be provided, with the aim of enhancing understanding.

In some cases, it may be possible to allow the planning team or stakeholders to interact directly with the model in facilitated sessions (see Case Study 5). This can be an efficient way for planners and stakeholders to gain insight into the trade-offs that exist and their options for dealing with them. Facilitation is key for this approach to be effective.

Box 10.2. Constructing a Stock and Flow Model

The first step in building a stock and flow model is to create a conceptual diagram that describes the components of the system and their connections. Sometimes the entire system can be captured in a single model. In other cases, individual objectives are modelled separately. Fig. 10.4 presents a conceptual model for a woodland caribou system.

Stock and flow models are named for the two types of components they contain. Stock variables are things that can be counted, like caribou. Flow variables are things that can be measured as rates, like predation. Flow variables are what cause stocks to change over time.

For SDM applications, the outcomes of interest are represented as stock variables, and proposed management actions are represented as flow variables. If any of the management actions act indirectly, then the intervening components also need to be included. For example, in Fig. 10.4, wolf control measures are directed toward wolves, not caribou, so wolf density needs to be included as a stock variable. Wolf density is linked to caribou population size through a flow variable describing the rate of wolf predation.

Additional flow variables may be required to account for the effects of external processes, such as climate change (blue ovals in Fig. 10.4). Processes are considered to be external if they are not amenable to management control within the context of the decision frame. As with other forms of modelling, a balance must be sought between adding detail and maintaining overall tractability. In SDM applications, we are most concerned with processes that alter the relative performance of management alternatives, affecting our determination of which is best.

To use the conceptual model for making predictions, it must be parameterized, which means that values are assigned to all of the variables. Stock variables are initially set to the current state of the system—the starting point of our projections. Flow variables that describe management actions are defined by the management alternative being examined in a given model run. Other flow variables are treated as cause and effect relationships between stock variables and are generally defined using equations derived from empirical research. For flow variables that describe external inputs, values are typically based on past behaviour or the extrapolation of trends.

The last step is to convert the conceptual model into a computer simulation model that can be used to track outcomes. Computers provide the computational power needed to explore complex systems quickly and reliably. Once parameterized with the current state of the system, the simulation model can be used to explore how the system will evolve under alternative management approaches.

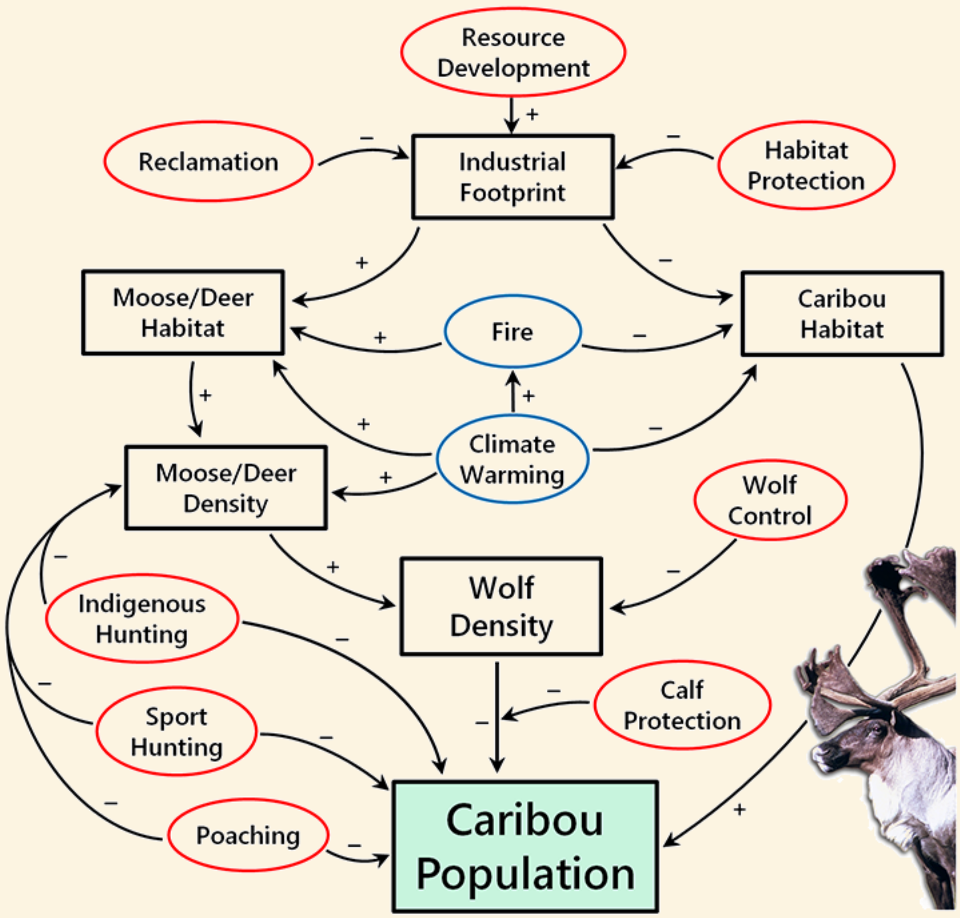


Fig. 10.4. A conceptual model illustrating the effects of management actions on woodland caribou populations. Stock variables are shown in black squares, flow variables for management actions are shown in red ovals, and external flow variables are shown in blue ovals. Plus and minus signs indicate a net positive or negative relationship. This model is a synthesis based largely on federal and provincial caribou recovery plans. Photo credit: J. Nickles.

Handling Uncertainty

Uncertainty presents a major challenge for decision makers. It is difficult to choose a preferred management alternative when the predictions of performance are unreliable. Therefore, every effort should be made to minimize uncertainty when obtaining data and constructing models. Residual uncertainty should be quantified so that it can be taken into account when decisions are made.

The methods for minimizing uncertainty when conducting observational studies are well developed. These are core elements of the scientific method (see Chapter 4). They include established techniques for designing studies and making observations, as well as statistical methods for generating robust predictions and quantifying the level of uncertainty.

Process models present a greater challenge (McCarthy 2014). Starting parameters are subject to measurement error and the cause-effect relationships among components are often only approximations. Biological systems are complex, and many modelled processes may be only partially understood. Extrapolation error is another problem, arising when information from external study areas is used. Finally, uncertainty arises from what is *not* in the model. The number of components that can be included is constrained by data availability, modelling capacity, and the need to maintain tractability.

The main tool for handling uncertainty in process models is **sensitivity testing** (Jackson et al. 2000). This entails running the model across a range of possible parameter estimates for each variable, rather than just using the best estimates. The range of values should include the outer bounds of what can reasonably be expected for each parameter, based on existing knowledge.

The sensitivity testing process usually begins by exploring the sensitivity of one variable at a time, in sequential model runs. Thereafter, logical combinations of variables may be explored together, up to the limit of what is practical. The number of permutations (and model runs) needed for combined sensitivity testing quickly becomes unmanageable when more than a few variables are involved.

It is usually more efficient to begin with a simple model, and then add detail where it is needed to reduce uncertainty, than to begin with a highly complex model. Moreover, only variables that contribute to the discrimination of management alternatives warrant inclusion.

For processes that are very poorly understood, the role of uncertainty can be investigated by asking "what if" questions—another form of **scenario analysis** (Peterson et al. 2003). Climate change presents an obvious example. There are still so many unknowns and contingencies related to climate change that we cannot treat the rate of warming as just another variable. Instead, we are faced with a number of plausible scenarios, with little guidance as to which is most likely. These scenarios can be investigated to determine how robust the model's predictions are likely to be, given alternative climate futures.

Further reduction of uncertainty can be achieved through scientific study, which mostly occurs between planning cycles. This is the tie-in to our discussion of policy-relevant research in Chapter 4. The contribution of SDM is that it identifies and focuses attention on the uncertainties that matter most to management. Research and learning can also be integrated into the decision process itself—a topic we will turn to at the end of the chapter.

A core principle of SDM is that it is better to make timely decisions on the basis of the best information available than to wait for better information to arrive (Martin et al. 2009). This is particularly true for biodiversity objectives, where delay in decision making generally equates with continued decline (Gregory et al. 2013).

A related principle is that all decisions should be seen as provisional. As new information comes to light, and as conditions change, the balance among management options may very well change. This is why the SDM process is depicted as a cycle.

Expert Opinion

Many planning initiatives do not have the time or capacity needed for modelling. They rely instead on expert opinion for predicting the outcomes of management actions. Such predictions are usually qualitative rather than quantitative, and there are few options for characterizing the effects of uncertainty. Moreover, there is less transparency concerning the factual basis of the predictions. Nevertheless, this approach is often used because no other options exist.

When predictions are based on expert opinion, it is best to solicit input from a diverse group of experts. This is because specialized knowledge is typically narrowly based. To obtain information that is both detailed and complete, multiple perspectives are needed.

Consideration should also be given to how expert opinions are elicited. A structured approach is best, involving a combination of individual meetings and group workshops (MacMillan and Marshall 2006). The necessary context should be provided, and the questions should be well framed. It should also be made clear that professional opinions are being sought, not value judgments. The role of experts is to help decision makers understand the consequences of management alternatives, not to tell them what to do.

Workshops are particularly useful in that they allow experts to share perspectives and challenge each others' ideas. The objective is to document areas of agreement and disagreement, rather than to force a consensus. This is how uncertainty can be characterized when modelling is not an option. In addition, experts should be challenged to provide the reasoning underlying their predictions and to express their level of confidence in them.

Box 10.3. The Precautionary Principle

The precautionary principle states that measures to prevent environmental degradation should not be delayed because of a lack of full scientific certainty (UN 1992). The Canadian *Species at Risk Act* applies the precautionary principle in the context of preventing the loss of wildlife species (GOC 2002). When making resource management trade-off decisions, this principle directs us to err on the side of caution when there is uncertainty about environmental outcomes. The aim is to minimize the risk to biodiversity that such uncertainty poses.

In SDM, the precautionary principle serves as a value position rather than a decision-making rule (Gregory and Long 2009). Prompt management action and the minimization of environmental risk are objectives we would like to achieve to support environmental values. But in most real-world settings, environmental objectives do not automatically override other social objectives. A balance must be sought, which is why SDM is needed.

Traditional Ecological Knowledge

Traditional ecological knowledge (TEK) is another form of expertise that can be used to support decision making. TEK derives from the close association that Indigenous peoples have with the land. It is place-based and multifaceted. Usher (2000) describes four distinct components of TEK:

1. Factual knowledge about the environment, including personal observations, personal generalizations

based on life experience, and traditions and teachings passed down from generation to generation. This category ranges from specific observations to explanatory inferences about why things are the way they are.

2. Factual knowledge about past and current use of the environment, particularly as it pertains to the rights and interests of local Indigenous communities.

3. Culturally based value statements about how things should be and ethical statements concerning appropriate behaviour toward animals and the environment.

4. A culturally based worldview that underlies the first three categories, providing the lens through which observation and experience are processed.

TEK informs the value-based positions that Indigenous groups advance as participants in decision-making processes. It can also be used as a form of expert opinion for predicting the outcomes of certain types of management actions. In the latter case, it is necessary to disentangle the value aspects of TEK from the factual aspects. Under SDM, values inform objectives and choices, but predictions of outcomes are to be made as objectively as possible.

The main strength of TEK as a source of information is that it is obtained through an intimate and ongoing relationship between Indigenous people and the land. This permits detailed observations to be made over large areas and over long periods of time, especially about animal behaviour, population sizes, habitat preferences, and animal responses to human activities (Gadgil et al. 1993). In contrast, scientific researchers face capacity constraints that force them to choose between studying small areas intensively or large areas coarsely. Furthermore, few studies last more than a few years.

The main shortcoming of TEK is that the observations of natural phenomena, while numerous, are generally not structured or recorded (Usher 2000). Furthermore, the information is distributed among a large number of observers. Findings may be shared through informal dialog, but are not formally synthesized. This presents a considerable logistical challenge for applying TEK in the context of SDM. Moreover, the reliance on memory and oral communication limits the level of detail that can be obtained and makes it impossible to judge the level of reliability (Usher 2000).

There is a direct parallel here with the challenges of citizen science, discussed in Chapter 4. Naturalists have been recording natural phenomena for centuries. But it was not until people began making observations using smartphone apps (including a photo, time stamp, and GPS location) and uploading their sightings to centralized databases that the full potential of citizen science was realized.

In practice, Indigenous people and researchers generally do not observe the same things over the same time period. Therefore, TEK and scientific knowledge are often complementary. For example, a scientific telemetry-based study of Arctic foxes around Pond Inlet, Nunavut, found that individual foxes use terrestrial and marine habitats throughout the winter. This knowledge was largely inaccessible to local Indigenous people because individual foxes seen at different locations are not easily differentiated (Gagnon and Berteaux 2009). For their part, Inuit elders and hunters had TEK concerning fox diets and behaviour that scientists did not have the time or capacity to collect. A predictive model that combined both sources of information would be superior to one that relied solely on scientific information or TEK alone.

There are, of course, cases when TEK and scientific information are in direct conflict (Stirling and Parkinson 2006). This arises most often in the context of co-management, especially as it pertains to hunted populations (Nadasty 2003; Armitage et al. 2011). In such cases, the criteria we discussed earlier concerning information reliability should be applied. However, there is also a social dimension that cannot be ignored.

When decision making is shared between government and local Indigenous communities, the decision-making process is itself a point of negotiation (Houde 2007). It should not be assumed that SDM and scientific principles will automatically be supported by Indigenous communities. Support has to be developed over time through collective learning and the establishment of trust. This remains a work in progress. In the words of Natcher et al. (2005 p. 241), “co-management has more to do with managing relationships than managing resources.”

Identifying the Optimal Approach

Stakeholder-Based Selection

The methods used to select a preferred management alternative depend on the type of decision and the level of stakeholder involvement. We will begin by examining methods based on structured dialog, which are commonly used when stakeholders are directly engaged in decision making. We will then turn to quantitative methods, which are best suited to operational decisions that are technical in nature.

As previously discussed, stakeholder groups generally do not make final decisions; they provide recommendations to a decision-making authority. Organizations and individuals engage as stakeholders because they perceive an opportunity to advance their interests, or because they are concerned that not participating might lead to undesirable outcomes. While they may be willing to work collaboratively, participating organizations tend to maintain fidelity to their own objectives and priorities.

For decision makers, stakeholder engagement provides insights for navigating difficult social trade-offs. Engagement also improves the likelihood that the decisions will be accepted and successfully implemented. Consensus is a welcome outcome but is not the overriding objective. In the face of strongly divergent views, a forced consensus is likely to be superficial, leaving difficult trade-offs unacknowledged and unresolved (Gregory et al. 2013). More important is the learning that occurs throughout the process, which takes several forms:

- Clarification of desired outcomes
- Identification of potential management actions
- Elimination of unworkable approaches
- Insight into the pros and cons of the best available management alternatives
- Identification of points of agreement and fundamental disagreement

Under SDM, the selection of a preferred management alternative is the endpoint of an iterative process of refinement (Fig. 10.5). Stakeholders will have differing perspectives and can help each other understand the implications and significance of the assessment results. Through structured dialog, the group can arrive at a common understanding of the strengths and weaknesses of individual alternatives, providing the foundation for further refinement. Good facilitation and a well-designed process are essential for making progress.

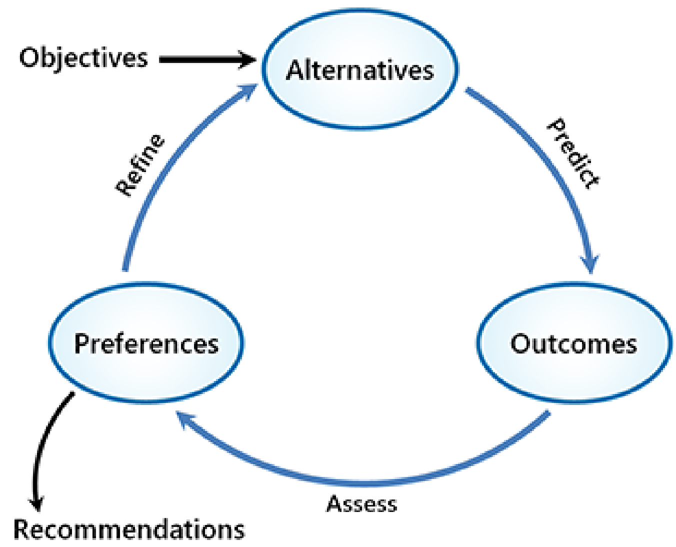


Fig. 10.5. The iterative cycle used to refine management alternatives and then select a preferred option.

In many cases, the significance of the predicted outcomes may require interpretation. For example, if our caribou population is expected to double under a certain management alternative, is it now within its natural range of variation? Or is it still below the minimal viable population size? Or something in between? Each of these connotes a different level of risk, and it is this risk that is of fundamental concern.

Another issue is the level of uncertainty associated with the predicted outcomes. A management activity that directly affects an outcome can usually be predicted more reliably than an activity that functions through multiple intermediate steps. When uncertainty is an issue, the consequences of being wrong need to be taken into account. For example, when working with a critically endangered species, preference may be given to an approach with moderate effectiveness and high reliability instead of an approach that is predicted to work better but is subject to higher uncertainty.

If uncertainty is found to be high across all alternatives, it may prompt a search for management approaches that are robust under varying conditions, as discussed in Chapter 9. These are often referred to as “no regrets” options (Johnston et al. 2010). A complementary approach is to implement management actions within an adaptive management framework (see below).

The refinement process begins with the elimination of alternatives that everyone agrees are inferior across all objectives. The focus then shifts to ameliorating critical weaknesses in the remaining alternatives, either by modifying some of the actions or by adding new elements. Refinement may also entail the development of hybrids consisting of elements from different alternatives.

In the face of direct trade-offs, improvement in one objective is often associated with reduced performance in a competing objective. However, this relationship need not be linear. Many management actions exhibit diminishing returns above a certain threshold. Therefore, exploring different combinations of critical management actions may reveal a win-win scenario, or at least an alternative that represents a clearly optimal balance.

Once the alternatives have been refined as much as possible (or practical), the focus shifts to making recommendations. As previously noted, the aim is to identify and document areas of agreement and disagreement concerning the short-listed alternatives rather than to force consensus. One approach is to have each stakeholder provide a qualitative assessment of each alternative, using categories such as “strong preference,” “acceptable,” and “strong opposition.”

The final recommendations provided to decision makers should convey the pros and cons of each of the short-listed options and describe where the support and opposition lies. The intent is to inform decision makers of the range of options available and the core trade-offs that differentiate them. If any of the alternatives emerge with universal support, they should be highlighted. If agreement cannot be achieved, then it is useful to articulate what the main stumbling blocks are and what it would take to overcome them.

On public lands, government decision makers are responsible for making the final decision, drawing on stakeholder recommendations as well as their political judgment. Stakeholder groups provide insights and a depth of understanding that are otherwise unattainable. But a group’s determination of the optimal course of action—if it can agree at all—may not represent what is best in terms of the broad public interest. A government decision maker, considering all viewpoints, may arrive at a different conclusion.

Of course, government decision makers are subject to their own shortcomings and there is no guarantee that their decision will necessarily best serve the public interest either (see Chapter 3). Sometimes, no decision is made at all. For example, stakeholder recommendations may be accepted but not implemented, or the decision may be deferred for more research. Common pitfalls leading to derailment of the process at this stage include:

- Lack of certainty, lack of consensus, high political or financial cost, fear of failure, or simple inertia—all of which reduce the likelihood of decisive action
- Lack of alignment between what planners and stakeholders have done and the purpose, scope, and constraints set forth by the decision maker
- Decision maker bias and dysfunction, including collusion with powerful stakeholders making an end-run around the planning process
- Barriers to implementation arising from policy collisions and other jurisdiction and integration issues
- Closure of a political window of opportunity, which can occur as a result of a change in government

Box 10.4. Lackey’s Axioms of Ecological Management

Drawing on his experience as a manager with the US Environmental Protection Agency, Robert Lackey (2006) published what he called the axioms of ecological management. These axioms, presented in abridged form below, provide useful insights into the political realities of environmental decision making.

- Environmental management tends to be a zero-sum game, meaning that policy choices usually result in winners and losers. Therefore, compromise is normally necessary to craft a workable policy or plan. Win-win solutions are rare.
- It may seem that the most important factor in decision making is weighing the total benefits against the total costs. But in most cases, the question of who receives the benefits versus who will bear the costs is most important.
- In the face of a trade-off, the most politically viable decision spreads the benefits to a broad majority,

- with the costs limited to a narrow minority of the population.
- Potential losers are usually more assertive and vocal than potential winners and are therefore disproportionately influential in decision making. Money and political access can also determine influence.
- Many stakeholders will cloak their arguments as science to mask their personal policy preferences.
- Calls for more research are ubiquitous in resource management debates. However, even with complete and accurate scientific information, most policy issues remain divisive because differences are invariably over values and preferences, not science and facts.
- The meaning of words matters greatly, and arguments over their precise meaning are often surrogates for debates over values.

Systematic Searches

An alternative to trial and error refinement is to search for the optimal combination of management actions systematically. If there are three or fewer outcome measures to be considered, the results can be displayed graphically (Fig. 10.6). When there are many outcome measures to be considered, it is usually necessary to combine them into a composite score to facilitate comparisons.

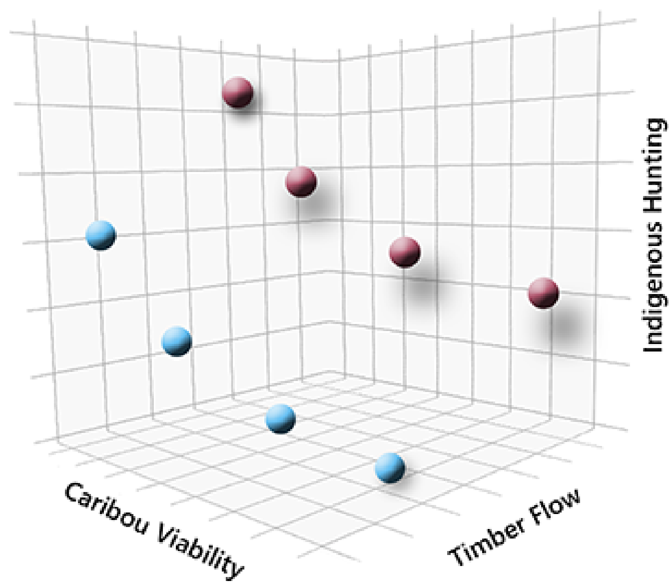


Fig. 10.6. The results of systematic explorations are best presented graphically. This hypothetical example involving caribou management illustrates a systematic search involving two types of actions across four levels each. The axes can be any three outcome measures of interest.

To generate a composite score, the outcome measures must first be standardized. The intent is to express the outcomes on an equivalent scale so they can be added together. In addition, a weighting factor needs to be applied to each outcome to account for differences in the importance of each objective. In practice, it is hard to achieve agreement on weighting factors within a stakeholder group because of the divergence of opinion about what is important. Therefore, these sorts of quantitative comparisons are usually reserved for technical types of problems tackled by planning teams that share a common perspective.

Fig. 10.7. Captive-reared whooping cranes being prepared for release. Credit: US Fish and Wildlife Service.

Converse et al. (2013) provide an example of systematic search methods applied to the recovery of whooping cranes (Fig. 10.7). Managers wanted to establish an additional crane population using captive-reared chicks, but the supply was limited, and chicks were also needed to support the recovery of the core crane population. An SDM process was used to determine the best course of action. The main objective was to maximize the probability of achieving a new self-sustaining population while minimizing the number of captive-reared chicks used. Additional objectives included cost, public relations benefits, and the value of the information gained from the reintroduction program.

The management options all pertained to the release of chicks into the new population. The options included (1) the timing and duration of the releases, (2) the number of chicks per release, and (3) a potential ten-year delay in the onset of the program. These options were systematically combined into 28 alternatives, representing all relevant permutations. An additional alternative was used to represent the reference case of no additional releases.

The predicted outcomes of each alternative were recorded in a consequence table, and a summary score was calculated for each (Table 10.2). This score was then used in making comparisons and selecting a preferred option (Fig. 10.8). To calculate the summary scores, the outcomes were all standardized to a 0–1 scale and then weighted to reflect their relative importance. For example, the population objectives were weighted more heavily than the public relations objectives. The weights were obtained from a team of conservation practitioners representing different organizations involved in whooping crane recovery.

Table 10.2. The consequence table used in the whooping crane reintroduction example.[1]

Objectives	Best	Worst
Population Viability[2]	0.289	0.122
Diverted Chicks[3]	30	50
Internal Cost (million $)	$9.95	$11.10
Partner Cost (million $)	$4.29	$2.86
Public Relations (0 or 1)	1	1
Information[4]	0.927	0.701
Weighted Score	0.657	0.320

[1]Source: Converse et al. 2013. Only the best and worst performing alternatives in the case study are shown.

2A weighted index of population viability based on multiple modelling methods.
3The number of chicks available for use in other reintroduction projects.
4Based on a formal value of information analysis.

When presented with graphical summaries like Fig. 10.8, it is important to recognize the influence of weighting decisions and the potential for subjective bias to creep in when defining the weights. Weighting decisions affect the ranking of alternatives, and as such, are integral to the determination of preferences.

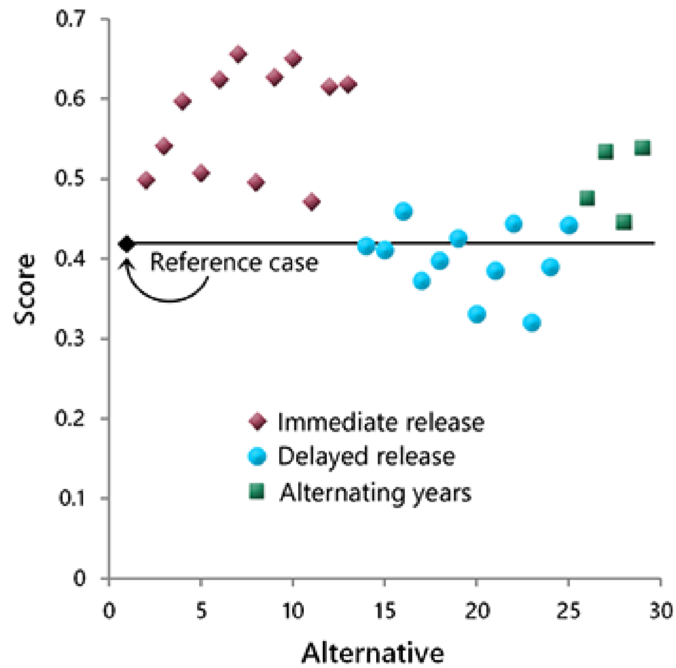


Fig. 10.8. The performance of all 28 management alternatives in the whooping crane example. The Score (y axis) is as described in Table 10.2.

In summary, systematic exploration provides an assurance that all combinations of potential actions have been considered. Having more data to work with also provides better insight into the nature of the trade-offs that exist. For example, it may be possible to determine whether a trade-off is linear or whether a break-point exists. The main shortcoming of this approach is that it is poorly suited to planning problems with many conflicting values and loosely structured alternatives. Had the whooping crane decision involved trade-offs with other species, or other social objectives, comparisons based on weighted summary scores would not have been appropriate. For these types of value-laden trade-offs, stakeholder-led dialog and learning is usually the preferred approach, despite its limitations.

Box 10.5 Mathematical Optimization

For certain well-structured planning problems, search methods based on mathematical optimization can be applied (Probert et al. 2011). In this approach, analytical techniques or computer-based optimization algorithms are used to identify the management option or combination of options that maximizes a selected objective (Rönnqvist 2003). Generally, only one objective can be optimized at a time, though other objectives can be included in the form of constraints. For example, the objective might be to maximize the volume of timber harvested, but a constraint could be added requiring the retention of a certain percentage of old-growth stands.

The main application of optimization methods in biodiversity conservation is in reserve design. As discussed in Chapter 8, programs such as Marxan can be used to identify the set of planning units that achieves representation targets at the lowest cost or with the smallest area. Few other conservation-related applications exist, mainly because of the difficulty in defining the mathematical objective function.

Implementation and Learning

Once a decision is made, the process enters the implementation phase. In some cases, particularly for lower-level decisions, planners will have direct control over the implementation process. This is an ideal scenario, in that there is high assurance that the plan will be implemented as intended.

More commonly, decisions take the form of rules or plans that are carried out by others. This is a major reason for including stakeholders in the decision process. Stakeholders can provide insight into implementation concerns, including feasibility, cost, and likelihood of success. This improves decision making and also helps to ensure that the alternatives can be implemented as specified.

Biodiversity Monitoring

For some SDM applications, a decision marks the end of the process. This is uncommon with conservation applications because threats to biodiversity are rarely fully resolved. Plans are developed, then periodically revised in response to new knowledge, changing conditions, and changing objectives. Thus, the implementation phase of one plan is also the preparation phase of the next plan (Fig. 10.1).

Learning is a critical component of the preparatory phase, and it occurs mainly through monitoring and research. We will begin our discussion of this topic with an examination of generic biodiversity monitoring programs. Then we will consider monitoring and research programs that are developed and implemented as part of individual SDM processes.

Generic biodiversity monitoring programs are concerned with tracking the overall status of biodiversity as well as threats to biodiversity (CCRM 2010). They are intended to reveal management successes as well as problems that require alternative approaches or additional effort. They are also intended to identify new and emerging concerns and bring these to the attention of managers and the public. Finally, the measurement of biodiversity within natural areas is used to establish ecological baselines and to characterize the natural range of variation.

The state of biodiversity at a given point in time is an integrative measure affected by natural and anthropogenic disturbances and by decisions taken at all levels of the decision hierarchy. The main indicators of biodiversity status at the species level are abundance and distribution over time. At the ecosystem scale, the focus is on ecological integrity, measured in terms of composition, structure, and function at multiple scales.

Biodiversity monitoring programs also track potential threats, especially industrial processes that cause disturbance or release waste into the environment. Tracking progressive changes in the overall anthropogenic footprint (i.e., cumulative effects) is of particular interest because this type of information is difficult to assemble from project-level monitoring. By combining long-term datasets on human activities with biodiversity trends, causal relationships that are critical for conservation decision making can be identified.

In practice, our ability to monitor biodiversity is severely constrained by funding limitations, institutional factors, and practical feasibility. Responsibility for wildlife management is fragmented, so there is no comprehensive, national-scale biodiversity monitoring program in Canada. The federal government oversees the monitoring of a few select species groups, such as migratory birds. It also conducts satellite-based monitoring of select biophysical attributes (Wulder 2011). Citizen-science programs, such as the Breeding Bird Survey, also contribute to national-scale assessments (Sauer and Link 2011). As for species at risk, status assessments are coordinated nationally, but the information used to make the assessments comes from diverse local sources (CESCC 2022).

At the provincial/territorial scale, biodiversity monitoring is highly variable. Monitoring programs are conducted at different spatial and temporal scales, measure different parameters, and use different protocols for data collection and analysis (CCRM 2010). The result is an information patchwork with many inconsistencies and gaps.

Focal species, including species at risk and species that are harvested, generally receive the most attention. Many of these species are regularly surveyed, providing ongoing estimates of abundance and distribution. The level of effort varies from province to province, depending on budget priorities and the perceived importance of a given species. Many species at risk are rare and/or difficult to census. Therefore, status assessments are often based on expert opinion rather than systematic surveys. The monitoring of non-focal species and ecosystem integrity usually receives a much lower priority, and is rudimentary in many parts of the country, especially in the north. Alberta's comprehensive, province-wide system of biodiversity monitoring merits special mention because it provides an example of what can be accomplished (Box 10.6). Unfortunately, in the 20 years since its conception, Alberta's monitoring program has not been emulated by other provinces.

The information collected through broad biodiversity monitoring programs is meant to inform the policy development process and provide context for lower-level decisions. However, this feedback loop is generally unstructured, partly because the policy development process is itself mostly unstructured, and partly because the information is scattered and difficult to assemble. In many cases, poor institutional linkages also hinder the process. For example, considerable effort is expended in monitoring ecological integrity within national parks; however, the linkages needed to apply this valuable baseline information on the working landscape are often lacking.

Box 10.6. The Alberta Biodiversity Monitoring Institute (ABMI)

The ABMI monitoring program uses a grid of 1,656 sites distributed evenly across Alberta at a 20 km spacing (Fig. 10.9). Currently, 150 sites are surveyed annually based on a budget of $13 million, which includes data analysis and reporting (ABMI 2017a). Terrestrial samples are taken across a 1 ha plot. Birds are individually counted at nine point-count locations, mammal presence is recorded along linear transects, and the presence of vascular plants is recorded in four sample sites. Aquatic samples are taken from the nearest wetland. Bryophyte, lichen, and mite specimens are collected in the field and later identified in a laboratory. Site conditions are also recorded during the site visit.

In recent years, automated cameras and acoustic recorders have been added to the sampling protocol (ABMI 2017b). Efforts are also underway to monitor rare animals and plants, which require specialized sampling methods.

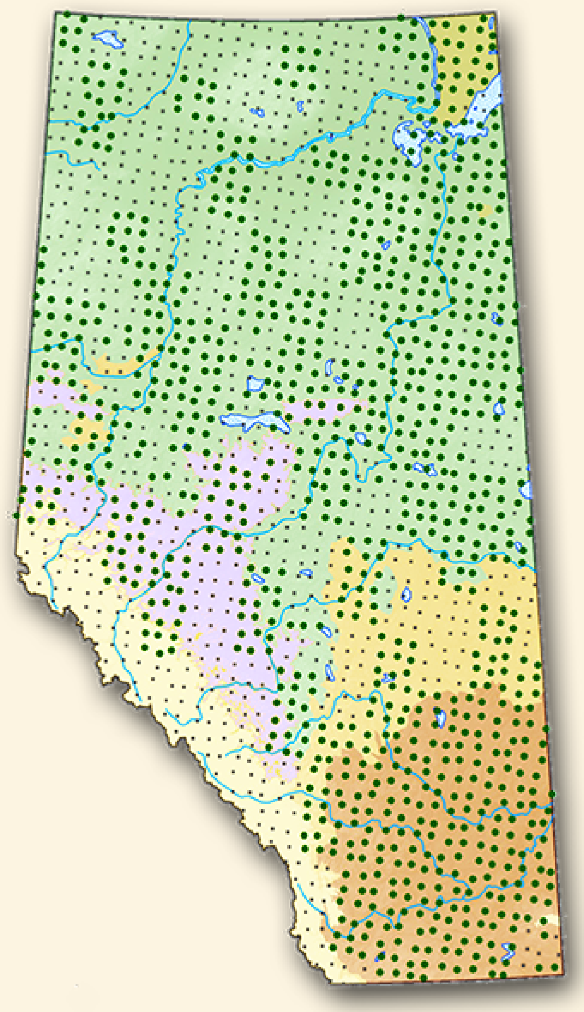


Fig. 10.9. The ABMI survey grid. The larger green circles have been sampled at least once (as of 2016) and the smaller grey points are awaiting their first visit. Source: ABMI.

In addition to the site sampling, AMBI uses remote sensing to document the human footprint, including disturbances related to agriculture, forestry, oil and gas, transportation, and human habitation. Fine-scale data (1:5,000) are collected annually within 3×7 km rectangles centred on each survey site, and coarse-scale data (1:15,000) are collected every two years for the entire province. ABMI also maintains a provincial-scale GIS database that includes detailed information on vegetation classes, soils, climate, and other landscape attributes.

An integrated science centre uses the collected data to develop models that relate species abundance to habitat information and human disturbance. With these models, province-wide interpolated maps of species distribution and abundance have been created for over 800 species. Other mapping products predict the species-level changes that have occurred as a result of development. Species and habitat data, mapping products, and analytical reports are all publicly available at no charge through an integrated web-based data portal (www.abmi.ca).

Outcome Monitoring

Monitoring is also conducted to support individual management plans. Indeed, under SDM, the acquisition of knowledge between decision cycles is a formal part of the decision process (Runge 2011). Using a structured approach for learning ensures that monitoring efforts are efficient and tailored to the specific needs of the plan.

The simplest and least expensive approach to structured learning is outcome monitoring. This entails tracking the objectives of a plan to determine whether the expected outcomes are being realized. This type of monitoring supports trial and error learning: if one approach fails, try another.

A noteworthy variant of outcome monitoring involves decisions with embedded **management thresholds** or triggers (Cook et al. 2016). In these applications, monitoring is used as a feedback mechanism for actuating predetermined management decisions. This approach is commonly used for managing air quality and water quality/ quantity. For example, water allocation rules may automatically change in the face of changing water supply (see Case Study 2). This approach permits management actions to adjust to changing conditions without having to wait for the next planning cycle.

Adaptive Management

Monitoring can also be applied as a structured learning tool to improve the predictive models that support decision making. This is referred to as adaptive management (Williams et al. 2009). Monitoring efforts are focused on the components of the model that have a large influence on the outcomes and are subject to high uncertainty. The intent is to fill gaps in knowledge and to validate the assumptions in the model, including implementation aspects. The overall aim is to improve the reliability of the predictive model, resulting in more informed decisions in the future.

Adaptive management is fundamentally a form of research. The hypotheses it seeks to test are the assumptions embedded in the predictive model. The study area is typically the planning area, though useful knowledge can also be gained from observations made elsewhere. For example, we might contrast observations made within the planning area, which we are perturbing through management actions, with observations made in natural areas.

Box 10.7. The Evolution of Adaptive Management

The adaptive management concept arose in the 1970s from frustrations resource managers had in applying models to real-world management problems (Walters and Hilborn 1978). Models were plagued by uncertainties that were inconvenient or costly for scientists to study because they involved processes that unfolded at large spatial and temporal scales. Adaptive management addressed this problem by using management actions as experimental treatments.

Adaptive management quickly gained adherents due to its inherent appeal. It was widely seen as a means to manage responsibly in the face of uncertainty. However, in practice, the adaptive management label was often applied indiscriminately to monitoring programs that did not feature structured learning based on predictive models. Furthermore, because adaptive management was initially conceived as a research tool, it proved difficult to integrate into the broader decision-making system, and this hampered its effectiveness (Walters 2007).

The adaptive management concept has evolved to address these shortcomings, and its role has been clarified. Today, it is seen as an integral component of the SDM cycle, rather than a stand-alone approach to research (Williams et al. 2009; Runge 2011).

Adaptive management initiatives can be differentiated into observational and experimental forms. Observational studies simply monitor what happens in response to the implementation of a preferred management alternative. In experimental studies, multiple management approaches are implemented in parallel as experimental treatments (Grantham et al. 2010). The observational and experimental forms of adaptive management are often referred to as "passive" and "active" forms, respectively (Williams 2011). However, these labels are not used consistently and are best avoided.

The appeal of the experimental approach is that it provides the fastest rate of knowledge gain. It also leverages the resources and management authority that are available to resource managers, enabling landscape-scale experiments that might otherwise be impossible to do. Experimentation epitomizes the concept of "learning by doing" that is central to adaptive management. Unfortunately, despite its appeal, there have been relatively few successful applications of experimental adaptive management (Walters 2007). In practice, it is difficult to secure the funding, staff, and institutional support needed for large-scale, long-term studies. Furthermore, it is often hard to obtain stakeholder support for experimentation in real-world settings.

An alternative to landscape-scale experiments is to conduct smaller-scale research studies focusing on specific processes. For example, forestry companies have long benefited from greenhouse experiments. Another option is to conduct pilot studies that involve just a portion of the planning area (Box 10.8).

Given the wide range of learning options available, the choice of which approach to use (if any) can be a difficult one. The decision should be based on a formal evaluation and comparison of the available options (Gregory et al. 2006). Detailed analysis is particularly appropriate when the management stakes are high and large sums of money are involved.

The main considerations are the costs, benefits, and likelihood of success of the various monitoring and research options (Gregory et al. 2006). Costs are measured in terms of the time and financial resources needed to implement the programs. Benefits are measured in terms of the ability of the acquired knowledge to improve future decisions. The assessment of the likelihood of success considers the ability of a learning program to deliver what it promises (Williams et al. 2009). The feasibility of implementing the program and the ability to generate statistically meaningful results are important factors. Institutional factors also need to be considered, including the level of commitment, capacity for sustaining a long-term program, systems for appropriately managing the data, and mechanisms for applying the findings to future decisions.

In addition to determining which learning approach will provide the best information for a given budget, there is the question of what the learning budget should be. This is a resource allocation issue that pits the benefits of learning against other management objectives (Gregory et al. 2006). As a general rule, learning should be emphasized when uncertainty about a decision is very high, particularly for issues of great importance (Fig. 10.10). There is little logic in implementing costly management actions when we are unsure of what actions to take. On the other hand, when the necessary management actions are abundantly clear, there is little justification for diverting funds to expensive research programs.

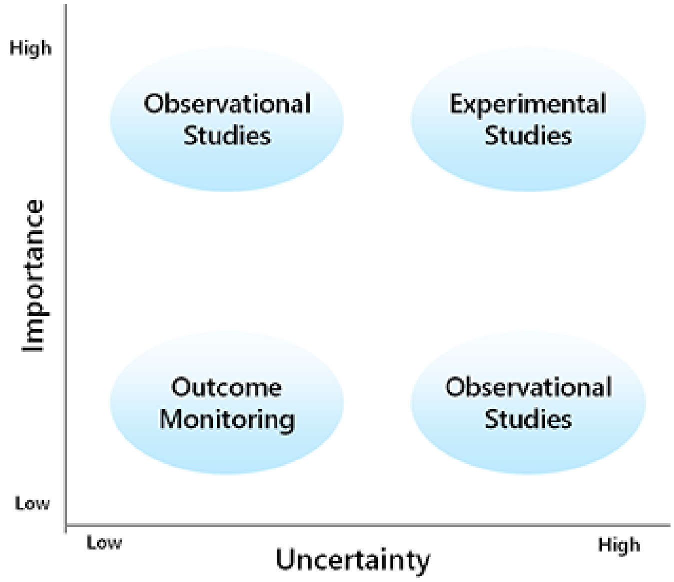


Fig. 10.10. The amount and type of learning should be matched to the level of uncertainty and the importance of the decision.

With all forms of monitoring, it is important to have a good information management system in place. Data should be consistently recorded, well organized, and easy to access. In addition, formal linkages should be in place to ensure that the collected data are incorporated into the next decision-making cycle. An effort should also be made to publish case study reports to facilitate learning in other areas.

Box 10.8. Learning Through Pilot Studies

Learning through passive monitoring is a slow process. Large-scale adaptive management experiments are much more effective but difficult to implement in practice. Pilot studies offer a middle road, providing a way to break complex problems into tractable components, and to test innovations at a small and manageable scale (Brunner and Clark 1997). These studies allow for flexible implementation in case of unexpected problems or opportunities. Indeed, an aim of all pilot studies is to devise a better program as experience is gained.

The small scale of pilot studies helps them maintain a low profile. This minimizes political visibility, and therefore vulnerability, until the results have been evaluated. If unsuccessful, a pilot study can be terminated more easily than a full-scale intervention because it is less likely to have acquired a large constituency willing to defend it. If successful, it can be expanded laterally and incorporated in the next decision cycle as a new and viable management option.

CHAPTER XI

CASE STUDIES

Case Studies

In this chapter, we will examine six case studies that illustrate how the conservation principles and methods described in previous chapters have been applied in real landscapes. The examples were selected for their instructional value and include both successes and failures. Each case study describes *what* happened and *why* it happened, with an emphasis on the decision-making processes and the role of conservation practitioners.

Each case study illustrates different conservation themes and involves different types of practitioners. Three studies involve ecosystem-level conservation and three involve focal species. A summary of the topics covered in each case study is provided in Table 11.1.

Table 11.1. Summary of case study topics.

1. Ecosystem management • Natural disturbance model • Triad approach • Integrated management	**4. Swift fox** • Species reintroduction • Multi-species action planning
2. Land-use planning • Regional planning • Cumulative effects management • Reserve planning	**5. Walleye** • Population modelling • Stakeholder engagement
3. Woodland caribou • Species at Risk Act • Conservation triage • Cumulative effects thresholds	**6. Protected areas** • Systematic conservation planning • Coarse filter vs. fine filter

Much of the information presented in these case studies comes from my experience as a participant or observer of the processes described. This is an inside view of how conservation works. Consequently, the examples are Alberta based. But to be clear, the location is not central to the themes that are explored; it is a backdrop. The emphasis is on people and processes, and understanding why things happen the way they do. The lessons learned have general applicability.

Case Study 1: Ecosystem Management

Background

This first case study recounts the development and implementation of ecosystem management by Alberta-Pacific Forest Industries (Al-Pac) in northeast Alberta.

The origins of this case study trace back to the late 1980s. The Alberta government was searching for ways to diversify the provincial economy, and forestry was identified as one of the main sectors for expansion. Pulping technology had advanced sufficiently by the 1980s to enable the cost-effective pulping of aspen. Moreover, pulp prices had risen to the point where building new plants was economically viable (Pratt and Urquhart 1994).

Operating under the assumption that there was a limited window of opportunity before pulp prices would again decline, the government acted quickly. In 1987, without public consultation or environmental study, it leased timberlands the size of Great Britain and negotiated the development of a dozen major wood processing facilities, including five new pulp mills. The bulk of this expansion occurred in northern Alberta, where vast tracts of forest were brought into industrial production for the first time.

The largest of the new forest management areas (FMAs) was acquired by Al-Pac, which was owned by a Japanese conglomerate. The FMA was 61,000 km^2, comprising almost 10% of the province (Fig. 11.1). To process all the new wood, Al-Pac proposed to build the world's largest single-line kraft pulp mill, supported by government loan guarantees (Pratt and Urquhart 1994).

What the government and Al-Pac failed to realize was that the social landscape was shifting beneath them. As we saw in Chapter 2, by the late 1980s, a societal tipping point had been reached, and the rules for managing public forests were being rewritten across the country. Al-Pac, as the company behind the largest of the new projects, became the focal point of public discontent and was targeted with massive and protracted protests.

The Alberta government refused to back down, and the mill was eventually constructed and began operations in 1993. Al-Pac emerged from its near-death experience as a unique entity. Simply put, Al-Pac went "green." Whereas existing companies and the government clung to conventional sustained-yield forestry, Al-Pac became a champion of new ecosystem-based approaches, turning its FMA into a research laboratory and proving ground for new ideas. It also implemented a chlorine-free approach to pulp production.

Fig. 11.1. The Al-Pac FMA, showing the location of the proposed Liege ecological benchmark.

Leading Al-Pac's new approach to forest management was an internal team of biologists and foresters. This group imported ecosystem management concepts from the US Pacific Northwest and then adapted them to Alberta's forests with the help of ecological consultants and academic scientists.

Our examination of Al-Pac's approach to ecosystem management will focus on the first decade of the company's operations, which is when the ecosystem management framework was developed and when most of the key decisions were made. Any significant changes that have occurred since that time will be noted.

The Natural Disturbance Model

Al-Pac's approach to ecosystem management centred on maintaining ecosystem attributes within the natural range of variability (NRV) through application of the natural disturbance model (see Chapter 7). In contrast to the Pacific Northwest, where ecosystem management led to a large decrease in the rate of harvest, Al-Pac intended to harvest at a conventional rate (and was required to do so under the terms of its lease). Working in Al-Pac's favour, the company had considerable operational flexibility because the forest had never been harvested.

Fig. 11.2. An aspen and white spruce mixedwood forest. Credit: R. Schneider.

In its application of the natural disturbance model, Al-Pac focused on four main attributes: forest composition, stand structure, landscape pattern, and age structure (Al-Pac 2007). The objective with respect to forest composition was to maintain the existing distribution of stand types. This required changes to the way that mixedwood stands, composed of aspen and spruce, were managed (Fig. 11.2). Under conventional practice, mixedwood stands were regenerated to either pure aspen or pure white spruce after harvest. This unmixing process progressively changed the composition of the forest (Hobson and Bayne 2000).

Al-Pac's solution was to implement an integrated approach to harvest planning referred to as "mixedwood management" (Lieffers and Beck 1994). The basic idea was to allow stands to regenerate as mixed stands and then harvest them twice: first to remove the fast-growing aspen—being careful to protect the spruce understory—and then later to remove the spruce, once it had matured. Because this approach was new, many knowledge gaps had to be addressed concerning growth and yield patterns, silvicultural techniques, and harvesting practices.

To implement mixedwood management, Al-Pac had to gain the support of several smaller lumber companies, referred to as quota holders. These companies had been awarded rights to most of the coniferous timber volume within the Al-Pac FMA (Al-Pac used mainly aspen in its mill). Because of their small size, quota holders were not on the public's radar and had experienced little pressure to adopt progressive practices. Motivating them to participate in mixedwood management was not easy, as they perceived risk with little reward. In the end, a decision was made to implement mixedwood management on a trial basis, on selected management units. Eventually, in 2010, these practices were extended across the entire FMA, in response to new forestry regulations.

At the stand level, Al-Pac's objective was to make harvest blocks as similar as possible to post-fire stands in terms of structure, size, and shape. Instead of the conventional practice of cutting square harvest blocks of uniform size (Fig. 5.10), Al-Pac's harvest blocks followed natural stand contours. Also, rather than piling and burning logging debris, as was common practice, logging debris was scattered over the site. This step had to be rescinded in later years because the government believed that the retention of debris created an increased risk of fire.

To mimic fire skips (patches of unburned forest), clumps of live trees were retained within the harvest blocks (Fig. 7.8). Fire skips vary in size and shape because of the complexities of fire behaviour. Within the boundaries of a large fire, some stands may be virtually untouched whereas others may be completely burned. A study of eight large fires within the Al-Pac FMA found that, on average, 13% of small mixedwood stands (<10 ha) and 33% of large mixedwood stands (≥10 ha) remained as unburned patches (Smyth et al. 2005). Constrained by mill requirements, Al-Pac chose 5% as its live tree retention target—substantially below NRV but substantially greater than conventional practice (i.e., zero). Quota holders would only agree to a 1% retention target for their stands, though in later years the government required them to adopt a 3% target.

At the landscape scale, Al-Pac sought to replicate the broad spatial patterns created by fires. The challenge was that harvest block size could not exceed 500 ha because of public opposition to large clearings. This was far smaller than the large fires responsible for most of the area burned. Al-Pac's solution was to implement an aggregated harvest system that focused harvesting in large "disturbance units" over a 10–20 year period (Carlson and Kurz 2007). Over time, individual harvest blocks would coalesce into large areas of young forest, approximating the effect of a large fire.

For logistical reasons, and because of uncertainty over social acceptance, Al-Pac limited the size of disturbance units to between 1,000 and 30,000 ha. This represents a subset of NRV. Historically, fires less than 1,000 ha accounted for 6% of the area burned in the FMA, and fires greater than 30,000 ha accounted for 41% of the area burned (Al-Pac 2007). Nevertheless, it is substantially closer to NRV than conventional practice, which generates very small patches across the entire management area.

With regard to the age structure of the forest, Al-Pac's emphasis was on retaining old-growth stands, which were most at risk from forestry. Conventionally, older stands were harvested first, to avoid timber losses to fire or senescence. The long-term outcome was a truncated age distribution, with no stands older than the optimum age for harvesting (see Fig. 5.9). In contrast, Al-Pac sought to maintain at least 75% of natural levels of old-growth, in perpetuity (Al-Pac 2007).

The challenge was that the natural amount of old-growth was difficult to determine. Al-Pac believed that the existing amount of old-growth on the FMA was not representative of the long-term NRV. An unusual spike in burning had occurred at the turn of the twentieth century, and fire suppression had been practiced since the late 1970s, creating a larger than normal cohort of older stands in the 1990s. As an alternative, Al-Pac derived the NRV of old-growth from estimates of the natural rate of burning. The problem here was that expert opinion about fire occurrence on the FMA was divided. Estimates of the natural rate of burning ranged from 0.4% to 2.2% of the FMA per year—a fivefold difference (Cumming et al. 2000). Al-Pac selected an intermediate value of 1.3% and then used a timber supply model to predict the expected long-term average amount of old-growth by stand type (Fig. 11.3).

The modelled estimates of NRV were substantially different than the existing amount of old-growth on the FMA. In particular, the predicted mean for jack pine was three times higher than existing amounts, and the predicted mean for white spruce was almost three times lower than existing amounts (Fig. 11.3). The practical implication was that the majority of the existing old-growth white spruce on the FMA, which was harvested by the quota holders, was destined to be liquidated on the basis of Al-Pac's NRV targets. In light of the uncertainties associated with fire modelling, and the large discrepancy between modelled and existing amounts of old-growth, this was a high-risk strategy. Al-Pac has since done additional fire modelling, but the old-growth NRV estimates have not changed significantly (Al-Pac 2015).

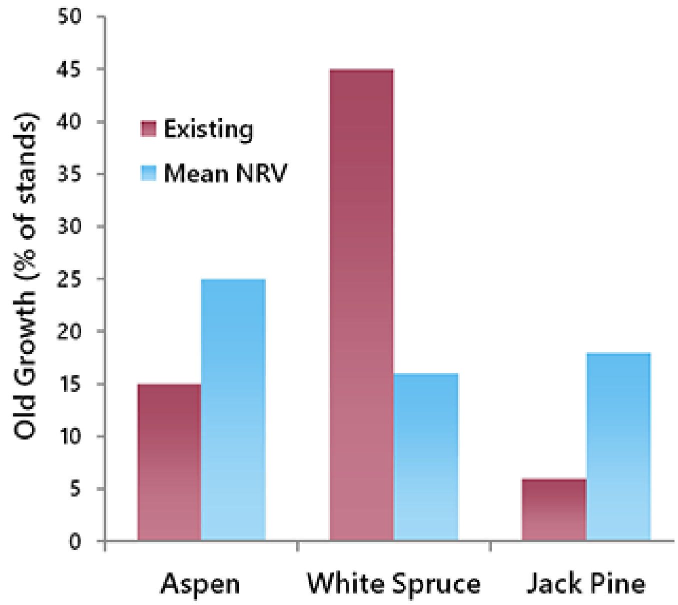


Fig. 11.3. The existing amount of old-growth on the Al-Pac FMA (red), and the predicted long-term average amount of old-growth on the FMA based on a burn rate of 1.3% per year (blue), by stand type. Source: Al-Pac 2007.

Another shortcoming of Al-Pac's timber modelling approach was that, following conventional practice in Alberta, it did not take future fires into account. The philosophy was that harvest levels could be adjusted later to accommodate fire losses. The concern with this approach is that future timber shortfalls may lead to increased old-growth harvesting at a later date.

The Triad Approach

The second pillar of Al-Pac's implementation of ecosystem management was the triad approach (see Chapter 7). The three zones of the triad included a sustainable forest management zone (most of the FMA), an unharvested ecological benchmark zone, and an intensive management zone. The site selected as the benchmark area was the 1,200 km^2 Liege River watershed in the northwest corner of the FMA (Fig. 11.1). This area was generally representative of the FMA and, notably, had not yet been allocated to either coniferous quota holders or the oil and gas sector. It was therefore relatively pristine and unencumbered.

For the intensive management zone, Al-Pac implemented an agroforestry program on privately owned lands immediately south of the FMA. In addition, the company instituted growth trials of fast-growing hybrid poplar species near the mill site (also on private land). The area of oil sands mining, in the central part of the FMA, was also considered an intensive management zone.

A critical shortcoming of Al-Pac's triad approach was that the company had no authority to enforce protection of the benchmark area it had selected, other than to curtail its own harvesting. Al-Pac lobbied the government to formally protect the site, but these efforts were not well received. Many within the government had a hard time understanding why Al-Pac would refrain from logging the forest it had just been allocated. Furthermore, even though quota holders and the oil and gas sector were not active in the Liege, they reacted negatively to the prospect of protection because it foreclosed future resource development options. The environmental community, for their part, supported the Liege, but only half-heartedly. The Liege had never been identified as a provincial priority, and the attention of environmental groups was focused elsewhere at the time.

In the end, raising the profile of the Liege watershed resulted not in its protection, but in its rapid leasing for oil and gas development. Al-Pac eventually abandoned it as a potential benchmark. It would take until 2018 for a large protected area to finally be established within the FMA, through a government-led land-use planning initiative (see Case Study 2). As for Al-Pac's experiment with hybrid poplar plantations, this initiative was abandoned in the 2010s for economic reasons.

Integrated Landscape Management

The Al-Pac FMA was a busy landscape in the early 1990s, home not only to Al-Pac, but also to oil sands companies, conventional oil and gas companies, coniferous quota holders, peat miners, and gravel miners. Collectively, the oil and gas sector cleared nearly the same amount of forest each year as Al-Pac (Al-Pac 2007). Yet these companies were not required to reforest the areas they disturbed—most sites were simply replanted to grass. Clearly, Al-Pac's vision of maintaining natural forest structures and patterns could not be realized without the support and participation of the other companies operating within the FMA. Thus, integrated landscape management became the third pillar of Al-Pac's implementation of ecosystem management.

Al-Pac's first step was to fully characterize the problem that needed to be solved. To do this, it enlisted the services of Brad Stelfox, an ecological consultant who had developed the ALCES cumulative effects model (www.alces.ca). ALCES was a bookkeeping model that tracked the state of a landscape as it evolved in response to disturbances (both anthropogenic and natural), as well as reforestation and succession. Using ALCES, Al-Pac showed how small disturbances from multiple operators would accumulate and fundamentally transform the FMA over the coming century. It also demonstrated how this industrial footprint could be substantially reduced through integrated planning (Fig. 11.4).

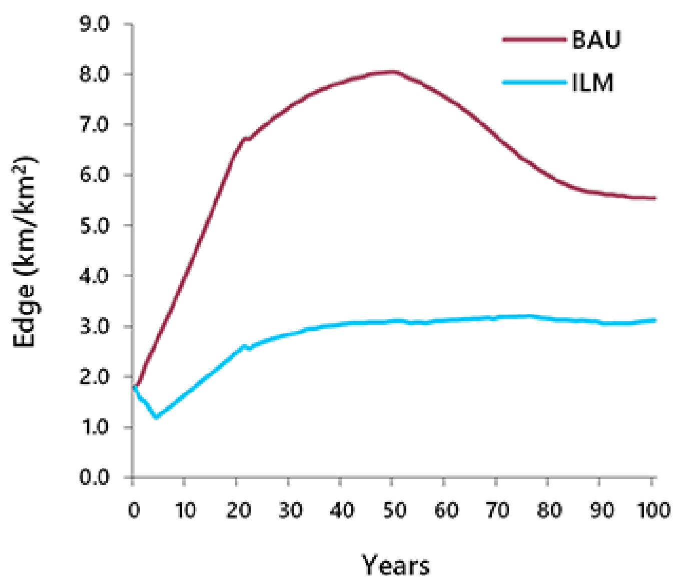


Fig. 11.4. ALCES model projections of the level of disturbance on the Al-Pac FMA over the next 100 years, measured in terms of the density of forest edge habitat resulting from roads, seismic lines, wells, pipelines, and harvest blocks. BAU = business as usual; ILM = integrated landscape management. Source: Schneider et al. 2003.

The second and harder step was to convince the other operators to cooperate. The coniferous quota holders were willing to have Al-Pac undertake joint harvest planning as long as their timber volume was maintained. But the oil and gas sector was not inclined to accede control of its planning to a forestry company.

Moreover, these companies had never been held accountable for their impacts on the forest, and the concept of maintaining ecosystem integrity was unfamiliar to them. ALCES model projections and moral suasion were helpful in changing attitudes, but only to a point. What Al-Pac needed was support from the government, but this was not forthcoming until the late 2000s (see Case Study 2).

In the absence of other options, Al-Pac pursued its integrated planning agenda through ad hoc company-to-company initiatives that could be presented as win-win solutions. One of the first projects was with Gulf Canada (now ConocoPhillips) on its Surmont oil sands development. Through joint planning of access roads, and by focusing harvesting in areas that would later be used for oil installations, the project minimized forest clearing, realized a 47% reduction in roads, and achieved more than $3 million in joint cost savings (Demulder and Thorp 2007). This set the stage for engagement with other oil sands companies. Al-Pac also encouraged seismic exploration companies to reduce the width of seismic lines by rebating their timber damage fees if they applied best practices (Moore et al. 2005). This led to a rapid reduction in the width of lines—from 5–6 m down to 2.5 m—and it showed how quickly and effectively industry could respond when motivated.

Decision Making

Two levels of conservation decision making can be discerned in this case study. At the top level was Al-Pac's corporate decision to pursue leading-edge forestry practices through the application of ecosystem management. This can be characterized as a risk-management strategy, motivated by the strong opposition the company encountered at the time of its establishment. Rather than remain a lightning rod for public discontent about forestry, executives decided that the company would fare better in the long run by becoming a leader in progressive forestry practices and mill operations.

The second level of conservation decision making concerned the myriad of operational decisions required to implement ecosystem management. Decision making at this level was led by the company's ecological team, composed of biologists and foresters. The working objective was to maintain forest attributes as close to NRV as possible, while accommodating mill requirements and other social objectives.

Initial efforts were focused on research. To maintain forest attributes within NRV, Al-Pac had to first determine what the NRV was. It also needed to understand how harvesting differed from natural disturbances and which differences were most important to biodiversity. Some research was done internally; however, Al-Pac also developed collaborative relationships with university researchers, leading to a period of intense study of Alberta's boreal forest. Hundreds of peer-reviewed studies involving the FMA were eventually published.

Al-Pac also engaged in the social aspects of conservation decision making. Initially, this entailed broad outreach efforts to raise awareness of ecosystem management and its benefits. Al-Pac also cultivated allies within the government, industry, academia, and stakeholder groups to help it advance its ecological vision and overcome resistance to change. Eventually, Al-Pac began to make the trade-off decisions that would define its version of ecosystem management. For the most part, the ecological team made these decisions internally, though it consulted extensively with other parties, both formally and informally. Mill requirements for profitability were taken as fixed constraints.

The team approached conflicts between conservation objectives and other social objectives as problems to be solved, to the greatest extent possible. Thus, considerable effort was devoted to the development of innovative solutions, rather than simply choosing from among existing approaches. It would ultimately take more than a decade for the core components and targets related to ecosystem management to be determined, and further refinement would occur in later years. Stretched out as it was, the process bore little resemblance to structured decision making. However, the main components (objectives, indicators, alternatives, trade-off decisions, and learning) were all present.

Analysis and Conclusions

This case study illustrates that conservation on public lands does not necessarily follow a linear path from public values to government policy to implementation measures. In this case, conservation was advanced through the direct influence of the public on a specific forestry company. The provincial government was more of a reluctant follower than a leader of conservation.

This example also illustrates the limitations of direct public action. The public certainly has the power to effect change, but it cannot grapple with details and complexity. In this case, all the attention was focused on Al-Pac, whereas the smaller quota holders and the energy sector were largely ignored. Consequently, only Al-Pac felt compelled to adopt progressive practices, and the other operators presented a barrier to change.

As to what was ultimately accomplished, there are three perspectives to be considered: the implementation of ecosystem management, the broader impacts on conservation, and biodiversity outcomes.

The assessment of Al-Pac's implementation of ecosystem management depends on the frame of reference used. If we use the original descriptions of ecosystem management in the ecological literature as our reference (e.g., Grumbine 1994), then Al-Pac's implementation falls short of the mark. This was not a "nature-first" approach, as was implemented in the US Pacific Northwest (MacCleery 2008). The mill's requirements came first, as did the needs of the quota holders and the energy sector. Consequently, only some forest attributes will remain within NRV as time progresses, despite the best efforts of Al-Pac's ecological team. The elements that are destined to change the most, relative to the natural state, are fire skips, burned forest, old-growth white spruce, broad landscape patterns, and the overall level of fragmentation and human access.

Alternatively, if we use conventional sustained-yield forestry as our yardstick, then Al-Pac's ecosystem management efforts constitute a major advance in conservation. Al-Pac was able to channel diffuse public anger over the Alberta government's massive timber allocations into concrete and meaningful improvements in forest management. Although forest attributes will not all stay within NRV, they will remain much closer to the natural state than if conventional forestry practices had been implemented on the FMA.

Looking at Al-Pac's efforts through a wider lens, the company's impact extended far beyond the boundaries of the FMA. Not only did Al-Pac implement ecosystem management, it was also a high-profile champion for this approach and helped to shape the public conversation about forestry in Alberta. The importance of these advocacy efforts, led by the ecological team, cannot be overstated. By serving as a proponent for conservation, and providing a working example of what could be accomplished within Alberta, Al-Pac became an agent of change.

As such, it was not always well received, but it could not be ignored. With the support of other parties, attitudes concerning acceptable forestry practices in the province slowly shifted, and forestry policy was eventually revised. In addition, Al-Pac's struggles with overlapping industrial operators helped to jumpstart a provincial dialog about managing cumulative effects.

The ultimate test of Al-Pac's implementation of ecosystem management is whether it is actually maintaining biodiversity. This is a difficult question to answer, despite ongoing monitoring. Al-Pac has only been active for 25 years, and most stands are still awaiting their first harvest. Moreover, many species exhibit lag effects to habitat degradation. Therefore, the effect of Al-Pac's activities, in combination with the impacts of other industrial operators, may not be fully apparent for some time.

According to data collected by the Alberta Biodiversity Monitoring Institute (ABMI), at this early stage of industrial development, the abundance of most monitored species (n=684) remains near natural levels when averaged across the entire FMA (ABMI 2020). However, the level of biodiversity intactness is notably lower in the areas where most of the industrial disturbance to date has taken place (Fig. 11.5). In these areas, characterized by high levels of habitat fragmentation, adaptable species such as coyotes and white-tailed deer have increased in abundance, whereas species sensitive to disturbance, such as caribou and golden-crowned kinglets, have declined (ABMI 2020; Fisher and Burton 2018).

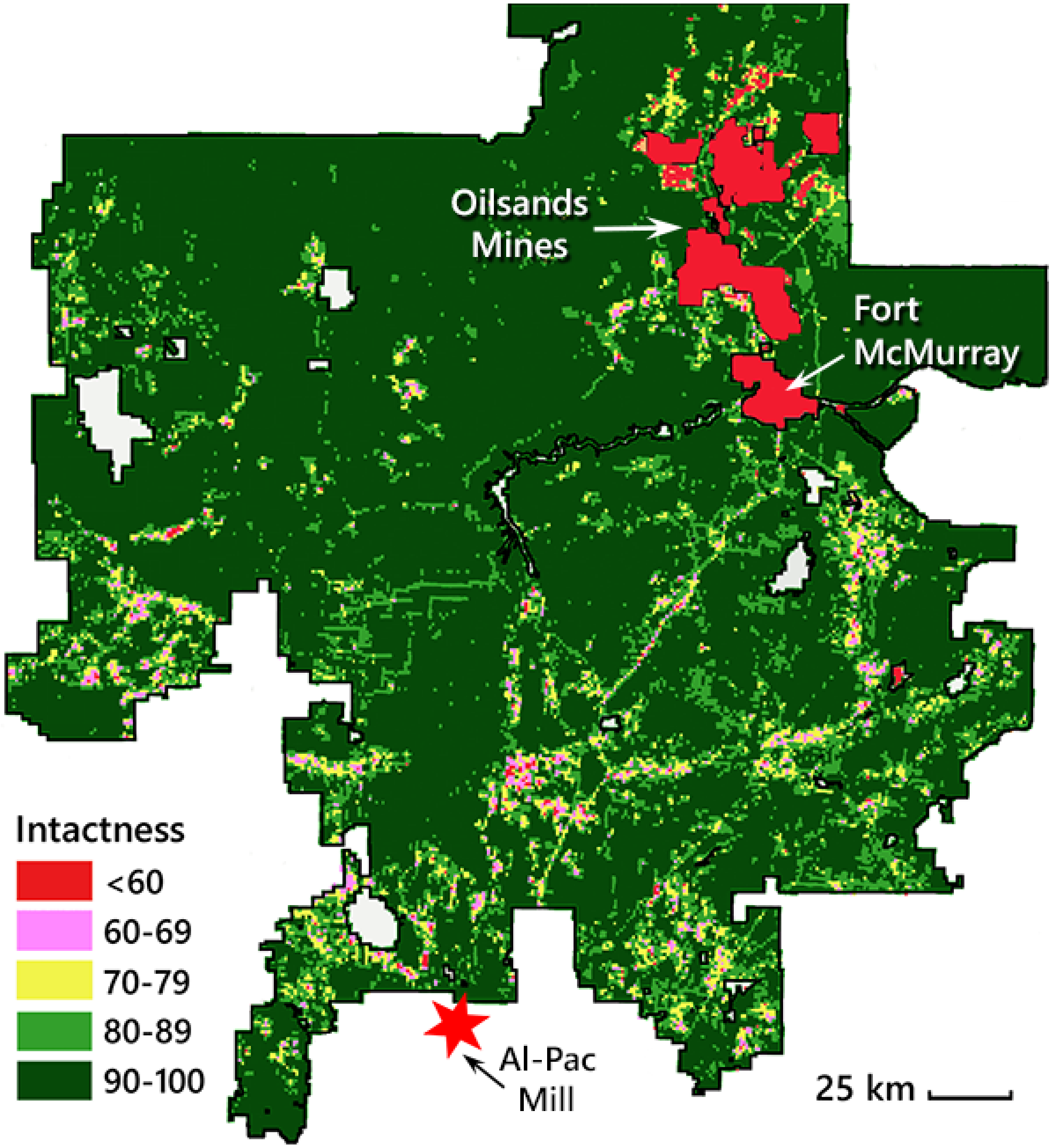


Fig. 11.5. A map of ABMI's biodiversity intactness index for the Al-Pac FMA in 2016 at the quarter-section scale. The results reflect the level of intactness averaged over 684 species. A value of 100 implies natural levels of abundance. Adapted from ABMI 2020.

In summary, the appraisal of Al-Pac's ecosystem management program demands a balanced view. Its innovative approach represents a significant advance over conventional forest management and contributes meaningfully to the conservation of biodiversity on the FMA. Yet, Al-Pac's version of ecosystem management must also be seen as a compromise solution that falls short of the ideal and is unlikely to maintain the natural abundance and distribution of all species. What the ecological team has essentially done is to optimize conservation outcomes given the constraints it was faced with. To achieve better conservation outcomes, these constraints must be addressed at a higher level of planning, which leads us to our next case study.

Case Study 2: Land-Use Planning

Background

This case study traces the development of the 2012 *Lower Athabasca Regional Plan* in northeast Alberta (Fig. 11.6; GOA 2012). Through this case study, we will explore the management of cumulative effects and the mechanics of government-led regional land-use planning.

The path that led to the *Lower Athabasca Regional Plan* began in the late 1990s with growing discontent over the government's laissez-faire approach to land management in the face of rising development pressures (Kennett 2002; Brownsey and Rayner 2009). Conflicts among land users were increasing and were not being resolved. There was also a perception that the government had abdicated its responsibility for the sound stewardship of public lands, as evidenced by the increasing degradation of Alberta's landscapes.

A key factor was the oil boom that began in the late 1990s, taking oil from under \$20 a barrel in 1998 to over \$100 a barrel in 2008. This rise in oil prices coincided with the first commercial application of steam-assisted gravity drainage (SAGD), a new extraction technique that enabled the recovery of deeply buried oil sands deposits. With this new technique, oil sands extraction became viable across ~140,000 km^2 of northern Alberta—an area almost the size of Florida—instead of the much smaller surface-mineable zone (Fig. 11.6).

Issues do not advance without advocates, and in this case, a network of conservation practitioners from several different organizations played a central role in propelling land-use concerns onto the political agenda. The organizations included the ALCES group (a consultancy), Al-Pac, the Canadian Institute of

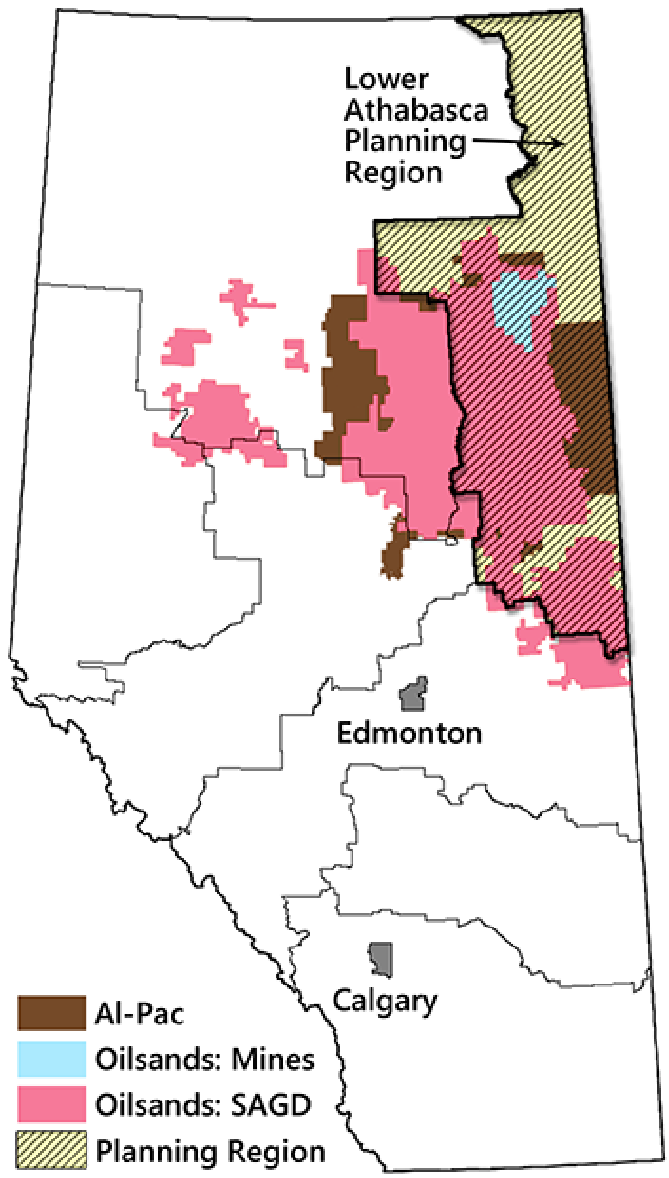


Fig. 11.6. The Land-Use Framework defined seven regional planning areas (outlined in thin black lines). The 93,000 km^2 Lower Athabasca Regional Plan area is highlighted in hatched yellow. The location of the Al-Pac FMA and the oilsands deposits are also shown (SAGD = oil recovery via steam-assisted gravity drainage).

Resources Law, the Edmonton chapter of the Canadian Parks and Wilderness Society, and the University of Alberta.

Collectively, this network helped to characterize the problem, devise solutions, and build support for change. Presentations that told the story of landscape change in Alberta—past, present, and future—were made to government officials and a wide variety of stakeholders, making the case for land-use planning. Members of the network also advanced ideas concerning the ecological, institutional, and operational aspects of managing cumulative effects (Kennett 1999; Schneider 2002; Weber and Adamowicz 2002).

Industry was also engaged, partly because conflicts between companies working on the same land base had to be resolved, and partly because of growing concerns about social licence. Leadership was provided by the Alberta Chamber of Resources, a cross-sector industry association. In 2000, the Chamber initiated a provincial integrated landscape management program that involved industry education, a series of pilot projects (similar to the Gulf Surmont project discussed in the previous case study), and a research chair at the University of Alberta (Demulder and Thorp 2007). Another notable development was the establishment, in 2000, of the Cumulative Environmental Management Association—a stakeholder forum for advancing the management of cumulative effects in the oil sands region funded mainly by industry.

The environmental community was just beginning to shift its focus from forestry to the oil and gas sector during the period of this case study. Their nascent efforts, which would later take the form of anti-pipeline protests, also provided impetus for addressing land stewardship concerns.

The Alberta Land-Use Framework

The initial government response to the growing concerns over land use took place within a single department: Alberta Environment. It established a new Integrated Resource Management Division in 2000, with a mandate to develop regional management plans. This effort proved to be unsuccessful, mainly because rivalries among government departments stymied progress (Brownsey and Rayner 2009). It became clear that integrated regional planning could not occur on the ground until the government itself became more integrated. This would require major changes in institutional structure, policy, and ultimately, political leadership (Kennett 2002).

The next attempt at integrated land management began in 2006, with the election of Ed Stelmach as premier. Stelmach named Ted Morton, a senior politician, as Minister of Sustainable Resource Development and charged him with developing a comprehensive framework for managing land use across the province. Integrated planning now had champions at the highest levels and could make headway against sectoral resistance. The resulting *Land-Use Framework* was released in late 2008 and incorporated input from several issue-specific working groups and extensive public consultation (GOA 2008). In 2009, the government passed the *Alberta Land Stewardship Act*, to provide a legal foundation for the Framework.

The *Land-Use Framework* outlined a new overarching approach for land management in the province. It explicitly acknowledged the land's finite capacity and the need for government leadership in coordinating development. The stated purpose was to manage growth in a way that would balance economic, environmental, and social goals (GOA 2008). Under the *Land-Use Framework*, the government would:

- Divide the province into seven regions (Fig. 11.6) and develop land-use plans for each. These plans would integrate provincial policies at the regional level, set out regional land-use objectives, and provide the context for land-use decision making within the region.
- Create a Land-Use Secretariat to lead the development of regional plans in conjunction with relevant departments.
- Implement cumulative effects management at the regional level.
- Establish an information, monitoring, and knowledge system.
- Include Indigenous peoples in land-use planning.

Of particular relevance to the conservation of biodiversity was the explicit commitment to manage cumulative effects. The *Land-Use Framework* required regional plans to define limits on the effects of development on the air, land, water and biodiversity of the region (GOA 2008). Within these limits, industry would be encouraged to innovate in order to maximize economic opportunity.

The Lower Athabasca Regional Plan

The first plan to be developed under the *Land-Use Framework* was the *Lower Athabasca Regional Plan*, in the heart of the oil sands region (Fig. 11.6; GOA 2012). Planning got underway in 2009 and was completed in 2012. The plan adhered to the direction provided by the *Land-Use Framework* and also incorporated concepts and strategies from earlier planning efforts by the Cumulative Environmental Management Association (the local oil sands stakeholder forum). Input from a regional advisory committee and the general public was also incorporated.

The Lower Athabasca Regional Plan defined seven regional outcomes and a set of strategies for achieving those outcomes (GOA 2012). The third outcome pertained specifically to conservation: "Landscapes are managed to maintain ecosystem function and biodiversity" (GOA 2012, p. 42). Most of the strategies in the plan were directives for further planning. For example, the plan specified that air, water, and biodiversity would be managed through a set of management frameworks that set targets for selected indicators and established triggers for proactive intervention. In addition, biodiversity objectives would be advanced by establishing new protected areas. We will examine the main biodiversity-related strategies in turn.

Protected Areas

The impetus for including new protected areas in the *Lower Athabasca Region Plan* can be traced to the *Terrestrial Ecosystem Management Framework*, developed by the Cumulative Environmental Management Association in 2008 (CEMA 2008), and before that, to a protected area campaign launched by the Canadian Parks and Wilderness Society in 2001. The additional protected areas were meant to fill gaps in ecosystem representation and to provide a better regional balance between industrial development and habitat protection. The Cumulative Environmental Management Association's consensus recommendation was that protected areas should comprise 20–40% of the region (CEMA 2008).

The main task for the Land-Use Secretariat was deciding where the new parks should go. To aid its decision, the

Secretariat commissioned a study by a group of researchers at the University of Alberta. The research group used the Marxan program (see Chapter 8) to identify reserve designs that achieved coarse-filter ecosystem representation while accounting for the contributions of existing protected areas (Schneider et al. 2011). The coarse-filter elements were derived from the provincial ecosystem classification system and a provincial vegetation inventory (NRC 2006). Preference was given to planning units with the lowest resource value and the lowest level of industrial footprint. Various permutations of reserve design objectives were explored (e.g., different levels of representation and different levels of reserve clumping).

The Land-Use Secretariat did not solicit public input about the protection options, and no public record exists for how the decision was made. The final configuration was broadly similar to a design submitted by the research group that used a 20% representation target and maximal clumping (i.e., 20% of each ecosystem type and each vegetation type had to be represented in a system of large contiguous reserves; Fig. 11.7). This was consistent with the low end of the Cumulative Environmental Management Association's protection target.

To be sure, the general configuration of the optimal reserve design was fairly obvious from the start. To avoid the high economic impact of curtailing oil sands development, new reserves had to be directed to the northern half of the planning region and along the Saskatchewan border. Another logical step was to use new reserves to connect Wood Buffalo National Park to nearby protected areas in a hub and spoke design (Fig. 11.7). The main contribution of the Marxan analysis was to demonstrate that such a design was capable of meeting ecosystem representation targets.

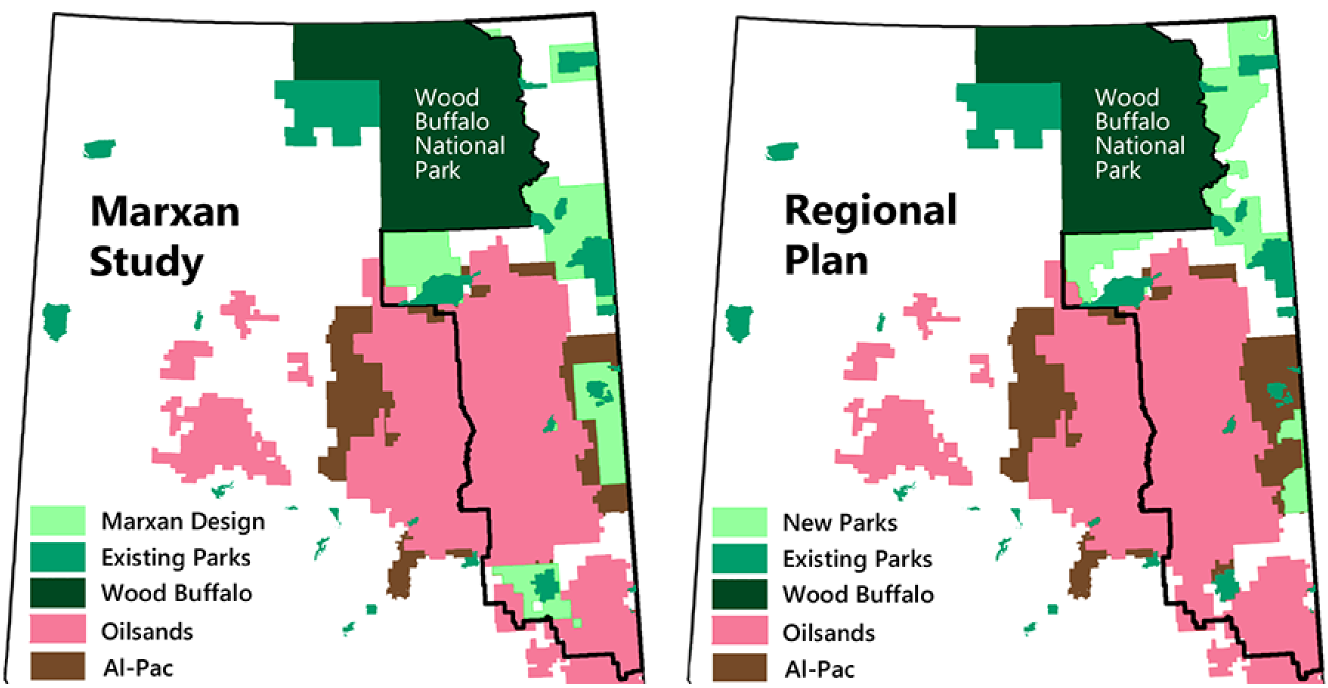


Fig. 11.7. The map on the left illustrates a reserve design generated by Marxan (Schneider et al. 2011). In this design, a minimum of 20% of each ecosystem and vegetation type was represented in the reserve system. Preference was given to planning units with the lowest resource value and the lowest amount of fragmentation. There was also a preference for large contiguous reserves. The map on the right illustrates the location of the new reserves that were established through the Lower Athabasca Regional Plan. The planning region boundary is outlined in black.

The reason Marxan was able to avoid selecting planning units within the oil sands region was that the surrounding landscapes were similar enough to be substitutable. This result hinged on the coarseness of the representation targets, which included only ecosystem type and major vegetation type. A finer-scale analysis would likely have uncovered features within the oil sands zone that are not found elsewhere.

The reaction of the resource industry to the new protected areas was mixed. Although existing oil leases were largely avoided, a few small companies were affected, and they were very vocal in their opposition. Most other companies understood the strategic value of establishing these new protected areas and were supportive.

As for the conservation value of the new sites, there are two perspectives. One is that the new reserve system is a major conservation achievement. A total of 13,600 km^2 of new protected areas were added to the existing system, bringing the level of protection within the Lower Athabasca Region to 21%. Combined with Wood Buffalo National Park (which lies just outside of the Lower Athabasca Region), these reserves constitute the world's largest contiguous boreal protected area, covering more than 67,000 km^2 (GOA 2018a).

A more critical perspective is that the newly protected lands contain almost no petroleum deposits, and so were never under substantive threat. Thus, there is no real conservation gain.

The reality is somewhere between these two views. While petroleum extraction is not a significant threat within the new protected areas, the sites do have potential for forestry and there have also been rising impacts from all-terrain vehicle use. Furthermore, experience suggests that new threats often emerge over time. For example, while mining is currently not an issue in the region, new diamond mining operations have started north of Wood Buffalo National Park.

Surface Water Quantity Management Framework

The *Lower Athabasca Regional Plan* called for the development of management frameworks for air, water, and biodiversity. In this section, we will examine the *Surface Water Quantity Management Framework* (henceforth the Water Framework), released in 2015. It provides a good example of the structured decision-making process in action (GOA 2015).

In this case, rather than conducting the planning process internally, the government enlisted the support of the Cumulative Environmental Management Association. A multi-stakeholder planning committee was established, and this is where most of the planning took place.

The scope of the Water Framework was restricted to the management of water quantity, and a separate framework was developed for managing water quality. Both frameworks focused specifically on the lower Athabasca River, the main waterway in the region. Narrowing the scope in this way made the planning processes tractable; however, it meant that water management across the broader region was not addressed. To date, no water management framework for the full watershed has been developed.

The objective of the Water Framework was to manage cumulative water withdrawals "to support human and ecosystem needs, considering an acceptable balance between social, environmental, and economic interests" (GOA 2015, p. 23). Clarity was brought to this broad objective by selecting indicators for three sub-objectives. For the economic dimension, the primary indicator was the volume of water available to oil sands companies for mining operations. For the environmental dimension, the primary indicator was habitat quality for fish. And for the social dimension, the primary indicator was river navigability for Indigenous communities during low-flow periods.

A hydrological model was developed by a technical team to help the planning committee explore trade-offs among the sub-objectives. The model predicted water flows under alternative management approaches while taking climate change into account. The committee iteratively refined management alternatives based on what they learned about trade-offs (Gregory et al. 2012, p. 232).

An important decision was to link the rate of permitted withdrawal to the rate of river flow (Fig. 11.8). This way, oil sands companies could store water on-site when it was plentiful, providing them with flexibility during low-flow periods. The committee was able to achieve consensus on limits for most flow scenarios, with the notable exception of 1-in-100-year low-flow events (which is when the trade-offs among desired outcomes were greatest). In this case, the government set the limit through an internal decision.

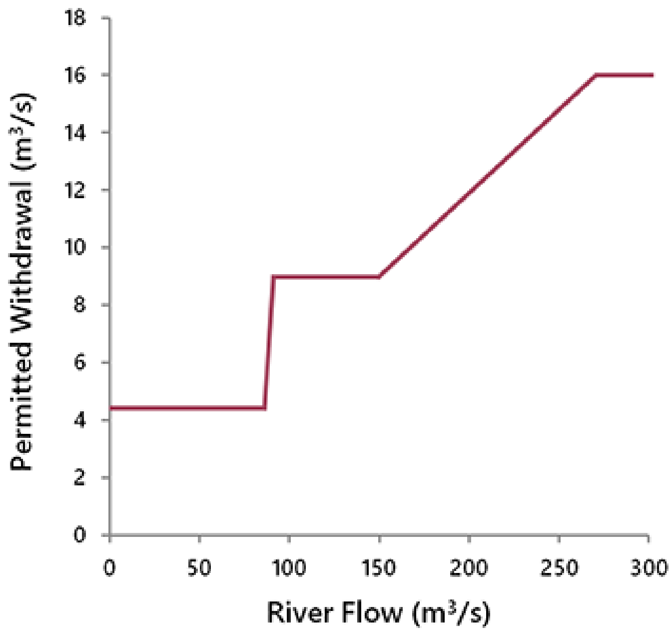


Fig. 11.8. Under the Surface Water Quantity Management Framework, water withdrawal limits are linked to the flow rate of the Athabasca River. The limits shown here are for Jan. 1 to Apr. 15. The limits in other periods are variations on this pattern. Adapted from GOA 2015.

The Biodiversity Management Framework

The *Biodiversity Management Framework* (henceforth the Biodiversity Framework) was due to be completed in 2013 but as of this writing (early 2023), it has still not been released. The description provided here is from a draft that was informally circulated in 2014.

In contrast to the Water Framework, planning for the Biodiversity Framework did not involve a stakeholder planning committee, even though the Cumulative Environmental Management Association had an active ecosystem working group and had released an ecosystem management framework of its own in 2008 (CEMA 2008). Instead, planning was done internally, through an interdepartmental planning team.

The Biodiversity Framework emphasized the broad goals of sustaining biodiversity and ecosystem health, carried over from higher-level plans. However, the specific meaning of “sustain” and “health” was not defined. Additional objectives included recovering species at risk, preventing new species from becoming endangered, and providing hunting and fishing opportunities for Indigenous communities.

It was not feasible to work with all species individually, so the planners devised a set of indicators that were meant to serve as biodiversity proxies. Four categories of indicators were identified: terrestrial habitat, aquatic habitat, terrestrial species, and aquatic species. For each category, a composite indicator was selected to represent the general state of biodiversity, and two indicators were selected to reflect specific biodiversity challenges in the region (Table 11.2). Additional indicators were used to provide supporting information, but they did not trigger management responses.

Table 11.2. Biodiversity indicators used in the draft Biodiversity Framework.

Category	Indicator
Terrestrial Habitat	Percent of upland area free of human footprint Amount of old-growth forest Percent of upland area that is at least 50 m from human footprint
Aquatic Habitat	Percent of wetland area free of human footprint Amount of undisturbed fen cover Stream connectivity
Terrestrial Biodiversity	Terrestrial biodiversity intactness index Woodland caribou Non-native plants
Aquatic Biodiversity	Aquatic biodiversity intactness index Arctic grayling Walleye

The use of a composite biodiversity index as a top-tier indicator is controversial. The benefit of using such an index is that it captures the status of the entire ecosystem in a single measure, which simplifies management. The drawback is that a composite measure may mask, through dilution, the very changes that conservation is intended to prevent. If the intent is to maintain biodiversity, then arguably, attention should be focused on the species most sensitive to disturbance (Devictor and Robert 2009). These are the weakest links in the chain.

In the Biodiversity Framework, each indicator was assigned an initial management response based on the level of risk it faced. Risk was determined by comparing the current status of an indicator with its reference state (i.e., its status when unaffected by human influences). Four levels of management response were defined, corresponding to increasing levels of risk:

1. **Low risk:** ongoing management
2. **Moderate risk:** improve knowledge and adjust management approaches as needed
3. **Considerable risk:** add new tools and shift from voluntary to mandatory requirements as needed
4. **High risk:** further increase in stringency, which *could* include restrictions on land disturbance and additional regulations

The management responses prescribed in this section of the Biodiversity Framework are too vague to be of any practical value—a major shortcoming. The only direction provided is that higher risk demands greater conservation action, which is self-evident. The question of what those actions should be is left unanswered. Nor does the

Framework provide any insight into how trade-offs with resource development objectives should be resolved. The only clear decision the planning team made was to demote the concept of cumulative disturbance limits to an optional management tool.

The land disturbance plan that was meant to accompany the Biodiversity Framework also foundered and there is no indication of when it might be released. Without clear direction on biodiversity objectives, the planning team charged with developing the disturbance plan struggled to make progress. Momentum for disturbance planning at the regional scale was slowly lost, and the emphasis eventually shifted to meeting the habitat intactness requirements of woodland caribou (see Case Study 3).

Analysis and Conclusions

This case study illustrates both the promise and peril of regional planning. Regional planning is needed to address the root causes of many conservation problems, particularly those related to the cumulative effects of industrial development. However, its inherent complexity makes it difficult to achieve substantive progress.

In this example, the political momentum attained in the late 1990s for addressing cumulative industrial impacts led to a paradigm shift in provincial land-use policy. The "open frontier" mentality that characterized earlier periods was replaced with an understanding that landscapes have finite capacity and that trade-offs among land uses have to be formally addressed. There was also a formal commitment to balance economic objectives with environmental and social objectives.

The effects this policy shift had on land management were varied. The most important achievement, from a conservation perspective, was the establishment of 13,600 km^2 of new protected areas. These reserves were intended to offset the impacts of development in the adjacent oil sands region. In addition, frameworks for managing regional air quality and water-related concerns in the Athabasca River were developed and adopted.

Much less progress was made in managing cumulative effects on the working landscape. Despite high-level policy commitments and more than 20 years of effort, no limit on disturbances has ever been implemented. As a result, the overall industrial footprint in the Lower Athabasca Region has steadily increased, despite the progress made through voluntary integration efforts (ABMI 2020).

Several factors contributed to the failure of the cumulative effects initiative. First, the scope of planning was not adequately contained. Instead of focusing on the management of cumulative effects, which it was originally intended to do, the *Lower Athabasca Regional Plan* became a catch-all for everything from economic diversification to providing recreational opportunities. This resulted in overwhelming complexity, superficial planning, and the eventual deferment of core planning issues (including cumulative effects) to secondary planning processes.

The initiative also struggled against internal government divisions and resistance to change, particularly from the proponents of economic development. Moreover, the governance system needed for integration at the regional scale was lacking. The emergence of Stelmach and Morton as champions for integrated planning provided a political window of opportunity for making substantive progress. However, this window did not remain open long enough for the required structural changes to occur.

Political factors were also important. A pivotal development was a disinformation campaign by the opposition Wild Rose Party, which turned the *Land-Use Framework* into a liability for the government instead of an asset. This campaign convinced many rural Albertans—a core constituency of the ruling Conservatives—that the government's real intent was to weaken their property rights. The recession of 2008 was another contributing factor, resulting in changing political priorities. Finally, the initiative lost its political champions when Morton changed ministerial portfolios in 2010, and Stelmach resigned as premier in 2011.

In 2012, when planning efforts finally turned to the management of biodiversity and cumulative effects, political attention had shifted elsewhere. Without high-level direction and support, the biodiversity planning team was in no position to begin setting strict regional limits on disturbance or even to launch a stakeholder-based planning process to investigate the options. Instead, the team spent the next six years in planning limbo, trying to establish a direction and ultimately producing nothing more than a draft system of biodiversity indicators. The cumulative effects "can" was kicked further down the road.

In principle, the legislation underpinning the *Land-Use Framework* should have ensured that the planning process was completed as intended. However, as with much of the environmental legislation in Canada, the *Alberta Land Stewardship Act* is discretionary. It provides the legal basis to develop and enforce regional plans but contains no explicit requirements to do so (Bankes et al. 2014).

The Water Framework provides a useful contrast. A structured decision-making approach was used to identify objectives, explore trade-offs among management alternatives, and make a decision on withdrawal limits. Part of what made this possible was the decision to narrow the scope to a single river. Complexity could not have been similarly contained for land-based cumulative effects because industrial activities are widely dispersed. Nevertheless, there is nothing about land-based cumulative effects that makes them impossible to manage given sufficient focus and political will.

In summary, this case study illustrates the wide range of factors that can affect regional planning, making outcomes difficult to predict. The importance of the problem being addressed and the level of support and opposition for change are fundamental factors. Internal government champions can make a big difference, but they come and go and windows of opportunity open and close. Regional planning may also become politicized, as evidenced by the Wild Rose Party's attack. Complexity is a major problem and can lead to delay and superficial planning. Finally, institutional barriers to integration pose a significant challenge. As a general rule, the greater the level of complexity or desired change, the higher the level of support and political momentum required.

Case Study 3: Woodland Caribou

Background

This case study explores the management of Alberta's woodland caribou (henceforth caribou; Figs. 11.9 and 11.10). As in most other provinces, caribou herds in Alberta experienced significant population declines and range contraction during the twentieth century. The fundamental causes of these declines and the management steps needed to recover the species were well established by the 1980s (Edmonds 1988). Overhunting and harsh winters were believed to be the initial causes, but later declines were attributed to the progressive industrialization of the forest (Edmonds 1988):

Fig. 11.9. Woodland caribou are found at low density throughout Canada's boreal forest. Credit: Peupleloup.

> Extensive timber harvest since the late 1950s has altered large areas of once occupied caribou habitat in west-central Alberta. Habitat of early successional stages developed, allowing for an increase in numbers and distribution of moose and, to a lesser extent, elk and deer. This increased prey base would, in turn, support a larger and more stable wolf population. Predation was the primary factor limiting the growth of caribou herds in our study area. … Unless immediate, intensive management is applied, caribou numbers and distribution in Alberta will continue to shrink. (pp. 825–826)

Industrial development also reduced the availability of preferred habitat and it facilitated human access into caribou range, which led to increased hunting, poaching, and vehicle collisions. The province's initial response was to curtail the sport hunting of caribou, in 1981, and to designate the species as threatened, in 1985. Wildlife managers also recommended habitat protection, a short-term wolf reduction program, law enforcement, and public education (Edmonds 1988).

Recovery efforts during the 1980s were led by provincial wildlife managers, who had little authority over the decisions that mattered. While these managers were recommending habitat protection, other branches of government were allocating vast tracts of caribou habitat for new industrial developments (as recounted in Case Study 1). Managers were able to achieve minor adjustments to harvest plans, such as the temporary avoidance of old-growth stands. But they had no success in permanently protecting habitat or reducing timber harvest rates (Hervieux et al. 1996). The proposed wolf control program was abandoned because of public opposition.

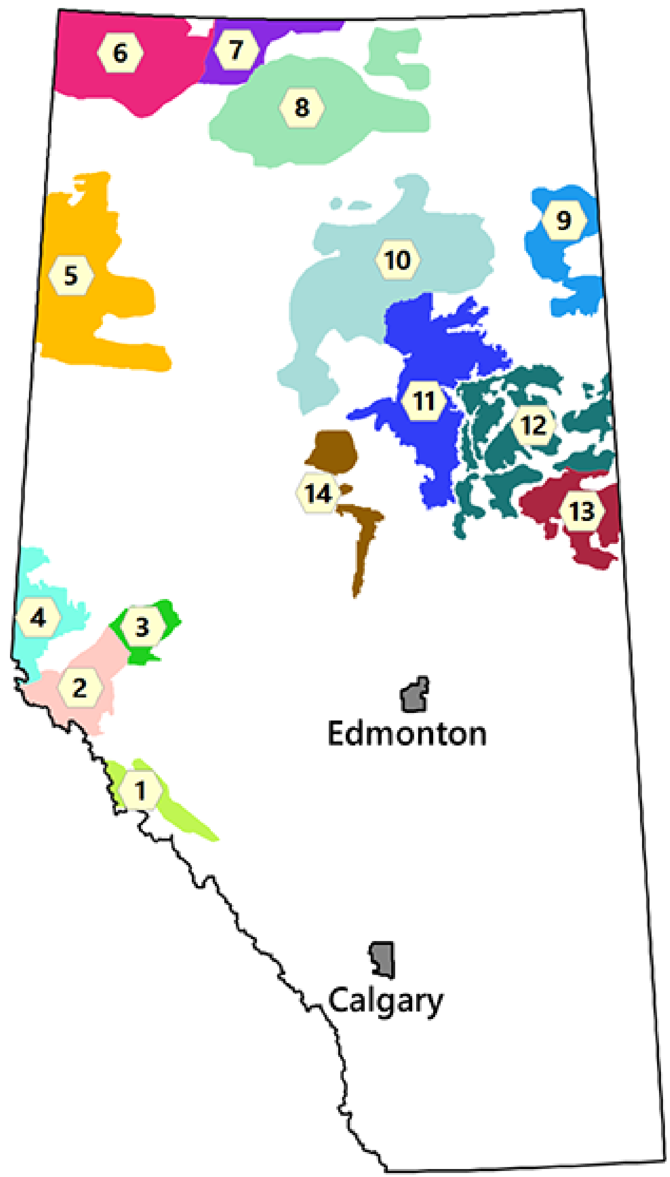


Fig. 11.10. Alberta's woodland caribou are distributed among 14 main ranges.

The 1990s saw the advent of stakeholder-based decision making, following the nation-wide trend that began after the War in the Woods (see Chapter 2). Caribou committees and working groups were established at the provincial, regional, and range levels. The stakeholders within these groups were roughly divided into two camps: those who sought substantive protective measures for caribou (government wildlife managers, conservation groups, and Indigenous people) and those who favoured the status quo (most resource companies and local communities). Unsurprisingly, these two camps could not find common ground. Moreover, elected officials were unwilling to intervene. This was during the period of laissez-faire land management discussed in the previous case study.

The ensuing period was characterized by relative stasis (except for caribou, which continued to decline). Every few years a new strategy or set of management guidelines would be released, but meaningful on-the-ground protection of caribou was not forthcoming (Hervieux et al. 1996). The only area of substantive progress was in research. The resource sector was unwilling to entertain constraints on development, but it was willing to provide funding for caribou field studies. There was a hope that such research would lead to win-win solutions that permitted resource extraction while also maintaining caribou. Research efforts could also be cited as evidence of conservation effort, offsetting the lack of demonstrable on-the-ground change.

Some research efforts clarified range boundaries. Other studies quantified population trends, particularly for northern herds that had not been well studied prior to the 1980s (Fig. 11.11). Basic caribou biology, including habitat associations and requirements, was also studied. Finally, there was an effort to refine and quantify the causal mechanisms underlying caribou declines. Many millions of dollars were spent conducting this research, making caribou one of the most intensively studied of all Canadian species.

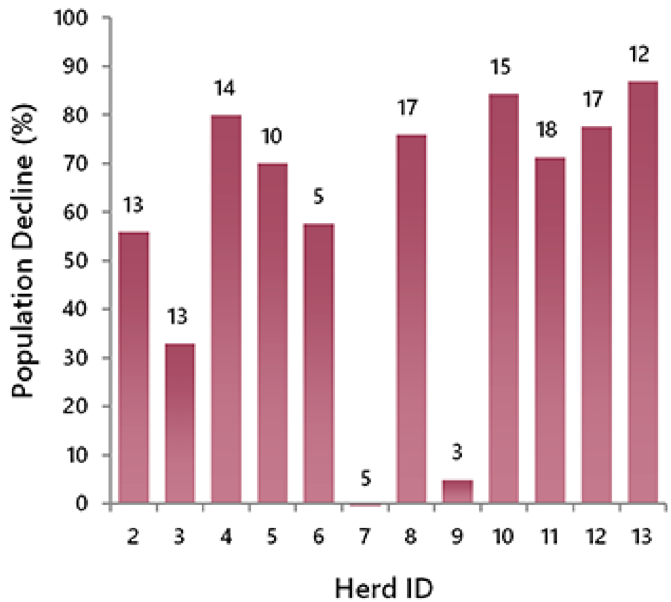


Fig. 11.11. The estimated decline of Alberta caribou herds over the period of active monitoring. The number of years of monitoring varies among herds and is shown above each bar. The herd ID corresponds to the labels in Fig. 11.10. Data are unavailable for herds 1 and 14. Adapted from Hervieux et al. 2013.

Triage

By the late 2000s, Alberta's caribou were acknowledged to be among the most threatened in Canada (EC 2011). Nevertheless, caribou committees remained stalemated. The proponents of development, including some branches of government, were unwilling to accept that caribou and industry could not coexist and balked at substantive habitat protection measures. For their part, caribou advocates were adamant that all herds in all regions had to be maintained, and this remained the stated goal of caribou management strategies (GOA 2011). The objective of maintaining all herds had the effect of concentrating management attention on the most threatened herds, which were now at risk of near-term extirpation.

A research group at the University of Alberta argued that the impasse could only be resolved through a compromise approach that acknowledged certain realities (Schneider et al. 2010). First, it was time to accept that caribou could not be maintained through minor adjustments of industrial operating practices. Caribou were uniquely susceptible to disturbance at the regional scale through the predation-mediated mechanisms articulated by Edmonds in 1988 (quoted earlier) and later verified by many other researchers (Latham et al. 2011; Peters et al. 2013; MacNearney et al. 2016). The accumulated evidence indicated that long-term caribou persistence required near-pristine conditions. Second, the economic cost of curtailing resource development and reclaiming the industrial footprint in all existing caribou ranges was too high to be politically viable (Hebblewhite 2017). Several herds occupied lands that contained resources worth millions of dollars per square kilometre (Schneider et al. 2010). Third, by focusing management attention on the most endangered herds, opportunities for the protection of other more viable herds were being neglected, placing their long-term viability at risk as well.

The research group proposed that conservation triage should be explored as a management option. To this end, they conducted an analysis that ranked the caribou herds in terms of viability and cost of recovery (Schneider et al. 2010). The group did not try to determine the number of herds that should be protected, as this was seen to be a matter of social choice. But they did argue that conservation efforts (i.e., full habitat protection and reclamation) should be allocated in a way that achieved the greatest overall benefit for caribou at the provincial scale.

In a subsequent study, the research group used Marxan to identify the best options for caribou habitat protection at the provincial scale (Schneider et al. 2012). First, habitat was assessed in terms of risk factors to caribou persistence, including the intensity of the industrial footprint and the potential for habitat transitions due to climate change (Fig. 11.12a–b). Next, the cost of protection was determined, expressed as the monetary value of resources that would become inaccessible after protection (Fig. 11.12c). Finally, Marxan was used to identify planning units that achieved specified levels of caribou habitat protection while minimizing habitat risk factors and cost (Fig. 11.12d). As it turned out, the spatial distributions of cost and risk were broadly similar, so the trade-off between them was minimal (compare panels a–c in Fig. 11.12).

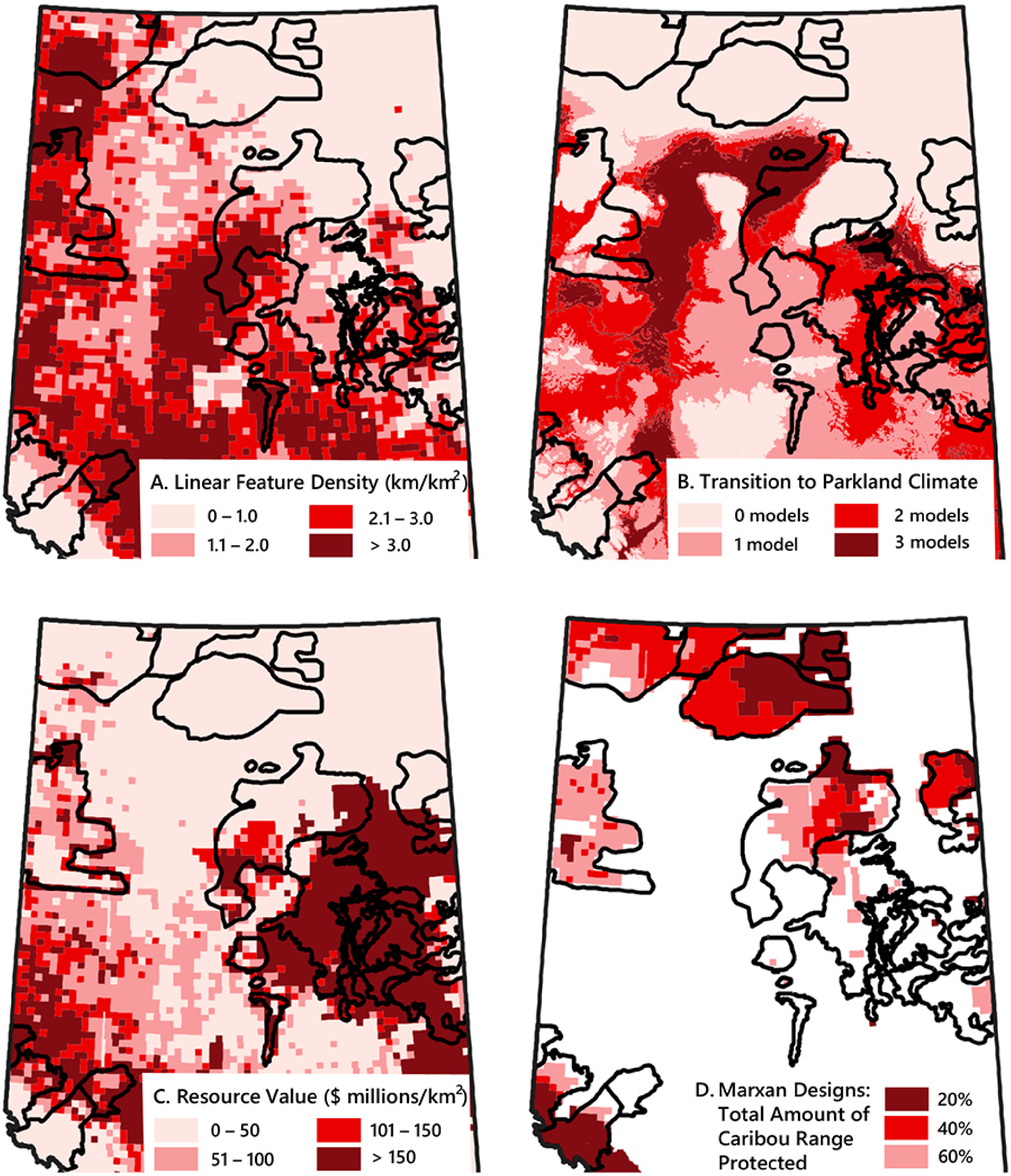


Fig. 11.12. The input and results from a study of caribou habitat protection options in Alberta. Map A shows the density of linear features, including roads, pipelines, and seismic lines. Map B shows the areas likely to transition to a parkland or grassland climate by 2050 according to predictions from three climate models. Map C shows the combined net present value of oil, gas, and forest resources. Map D shows the Marxan reserve designs for successively higher levels of caribou habitat protection (at the provincial scale). Most of the 20% protection target was achieved within existing protected areas. Adapted from Schneider et al. 2012.

The researchers found that, using optimization, 60% of current caribou range could be fully protected (including 17% in existing parks) while maintaining access to over 98% of the value of resources on public lands (Schneider et al. 2012). This was possible because most of the resource value in northern Alberta is concentrated in oil and gas deposits that only partially overlap with caribou range. The overlap with forestry is greater; however, the value of forest products accounts for less than 1% of total resource values. The prospects for protection were much reduced if protection was instead directed toward the herds that were most endangered.

The release of the triage studies had no discernible effect on caribou management in Alberta. Critics saw triage simply as the abandonment of difficult herds. They questioned whether the quid pro quo of protection and reclamation of the more viable herds would actually happen. There was also a concern that genetic differences among herds would be lost. As for the government, the optics of triage were highly problematic. Politically, there is a world of difference between quietly allowing herds to decline through neglect and publicly announcing that certain herds will be abandoned in favour of resource development, even if other herds will receive enhanced protection as a result. There was also the federal *Species at Risk Act* (SARA) to contend with, as it provided no latitude for triage.

The Federal Recovery Strategy

The boreal population of woodland caribou in Canada was listed as threatened in 2003, when SARA came into force. Under SARA, a federal recovery strategy was required by 2008. However, by 2011, the strategy had still not been completed and a coalition of conservation and Indigenous groups mounted a court challenge to spur the federal government into action (Ecojustice 2012b). When a draft strategy was released later that year, the government was inundated with over 14,000 public comments—an indication of the species' high profile (Paris 2012). The final strategy was released in 2012 (EC 2012c).

With the release of the federal recovery strategy, caribou management in Alberta (and other provinces) entered a new phase. Not only was the bar raised for recovery actions, but also, for the first time, these actions were non-discretionary.

The stated goal of the recovery strategy was to achieve self-sustaining local populations in all boreal caribou ranges throughout their current distribution in Canada. This goal effectively excluded triage as a management option. The justification for including all herds was that each herd contributed to population connectivity, redundancy, and the representation of local genetic adaptations, all of which are important for the long-term persistence of the species.

In terms of management actions, the recovery strategy was broadly similar to earlier strategies that had been developed in Alberta. It included habitat protection, wolf control, management of wolf prey species, voluntary restriction of Indigenous hunting, monitoring, and research. What differentiated the new federal strategy from all earlier efforts was the hard line it presented concerning the identification and management of critical habitat.

SARA defines critical habitat as "the habitat that is necessary for the survival or recovery of a listed wildlife species" (GOC 2002, Sec. 2). An expert panel, commissioned by the federal government, determined that the aspect of habitat which most affected caribou survival and recovery was the level of disturbance across the entire range (Fig. 11.13; EC 2011). For planning purposes, the panel developed a model that illustrated the relationship between total disturbance and the probability of maintaining stable or increasing population growth over a 20-year period (Fig. 11.14).

Following the expert panel's lead, the recovery team defined critical habitat in functional terms: within each range, a minimum of 65% of the area would have to be maintained in an undisturbed state. The 65% undisturbed habitat target corresponded to a 60% probability of herd stability (Fig. 11.14), which was judged to be an acceptable level of risk. The planning team reasoned that a 100% probability of stability would have been ideal but unrealistic, since "0% total disturbance is virtually impossible even without anthropogenic disturbances" (EC 2012c, p. 66). The provinces were given until 2017 to develop range plans that would describe how the 65% target would be achieved for each herd.

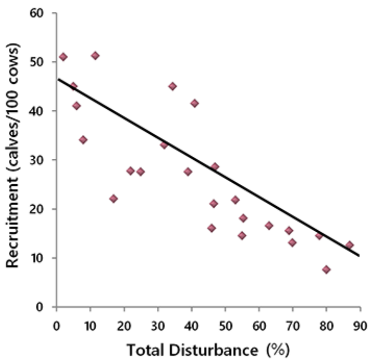


Fig. 11.13. There is a linear relationship between total range-wide disturbance and the rate of annual caribou recruitment (r^2 = 0.7). Disturbance is a composite measure that includes anthropogenic features, with a 500 m buffer added, and areas burned within the previous 40 years, with no buffer. The data are for 24 herds from across Canada. Adapted from EC 2011.

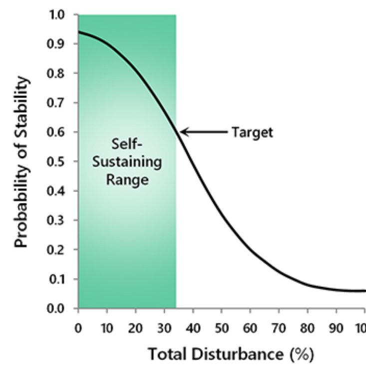


Fig. 11.14. The data presented in Fig. 11.13 can be used to show the probability of herd stability as a function of range-wide disturbance. The caribou recovery team equated critical habitat with a requirement to achieve a minimum 0.6 probability of herd stability (i.e., 60%). This threshold corresponded to a maximum 35% disturbance of the range (i.e., 65% undisturbed). Adapted from EC 2012c.

A subsequent federal policy document provided additional guidance for identifying and managing critical habitat for caribou (ECCC 2016c):

- Disturbed habitat includes areas of human disturbance, with a 500m buffer added around each disturbance, and areas that have burned within the previous 40 years, with no buffer
- In ranges with less than 65% undisturbed habitat, all currently undisturbed habitat should be protected from destruction
- In ranges with less than 65% undisturbed habitat, all areas possessing biophysical attributes for caribou should be protected from destruction
- In ranges with less than 65% undisturbed habitat, range plans should demonstrate how disturbed habitat will be restored to achieve the minimum 65% target, with timelines included

- Undisturbed habitat should be in contiguous tracts that facilitate connectivity
- Range plans should include a landscape management system that allows for ongoing disturbance (including fire) and renewal while ensuring that at least 65% of the area is always in an undisturbed state
- Range plans should identify the legally-binding instruments that will be used to prevent the destruction of critical habitat

The Alberta Range Plan

Alberta failed to complete individual range plans by the 2017 deadline. However, it did release a draft provincial range plan, which was meant to serve as a template for future herd-level planning (GOA 2017a). The draft range plan described how 65% of each range would be maintained in an undisturbed state through an integrated landscape management system. The centrepiece of the proposed system was a multi-use access network that would be developed for each range. According to the range plan, it would be possible, using spatial optimization techniques, to design a network that provided access to virtually all resources while still achieving the 65% caribou target.

The range plan also included aggregated forest harvesting, which we previously encountered in the Al-Pac case study. By concentrating harvesting in a specific area over a ten-year period, and then moving on to a new area, the overall level of disturbance on the landscape would be reduced (Fig. 11.15). The range plan contained no mention of altering annual forest harvest rates.

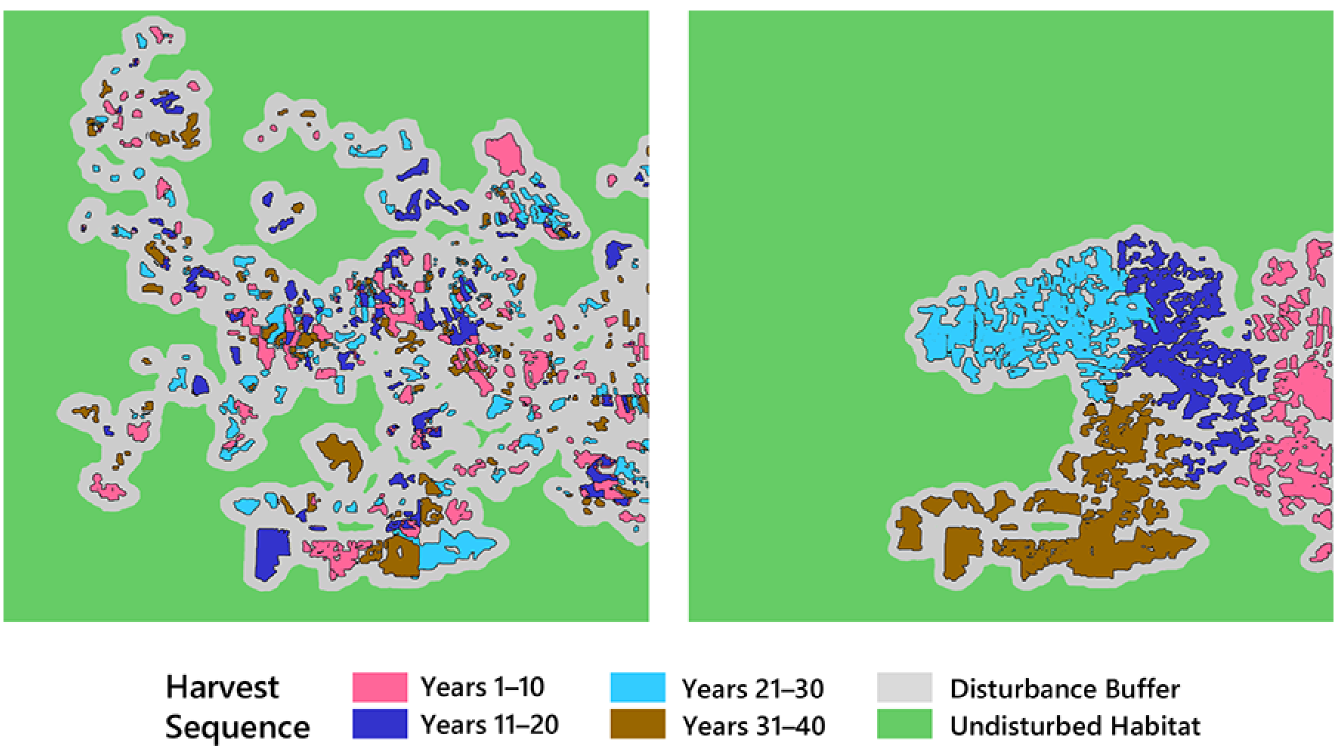


Fig. 11.15. A comparison of conventional (left) and aggregated (right) harvesting systems. In aggregated harvesting, cutblocks are concentrated in designated zones over ten-year periods, leaving a much greater proportion of the landscape undisturbed at any given time. Adapted from GOA 2017a.

The range plan acknowledged that achieving the 65% undisturbed target would require restoration efforts because existing levels of disturbance were very high in all ranges (Fig. 11.16). Under the range plan, restoration would be applied to industrial features that were no longer in use, particularly seismic lines and well sites. In addition, access routes would gradually be transitioned to the new optimized network. Given the extent of the existing footprint, it was estimated that it could take 50–100 years to achieve the 65% undisturbed target in all ranges (GOA 2017a).

The range plan also included measures for maintaining the viability of herds during the extended restoration period. The primary measure was wolf control, which was already being used in the highly compromised Little Smoky range (Herd 3 in Fig. 11.10; Hervieux et al. 2014). Additional proposed measures included the establishment of predator-free enclosures for calf-rearing and increased hunting of moose, deer, and elk to reduce prey availability for wolves.

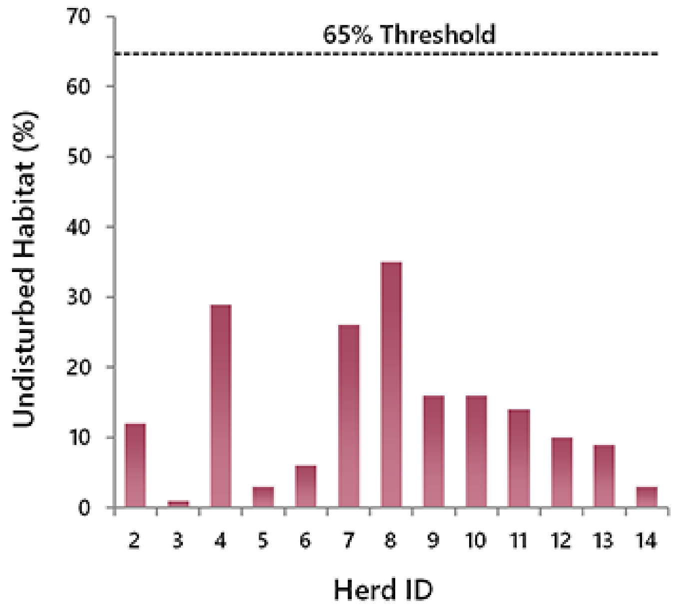


Fig. 11.16. The current amount of undisturbed habitat within each caribou range. The herd ID corresponds to the labels in Fig. 11.10. Source: GOA 2017a.

Finally, the range plan included protected areas as a recovery measure. Existing protected areas that overlapped with caribou range were incorporated, and the plan identified several new candidate reserves (Fig. 11.17). The new reserves were mainly the portions of caribou ranges that were undeveloped and held minimal resource value (Denhoff 2016). There was substantial overlap with the priority sites identified under the triage approach (compare Fig. 11.17 with Fig. 11.12d).

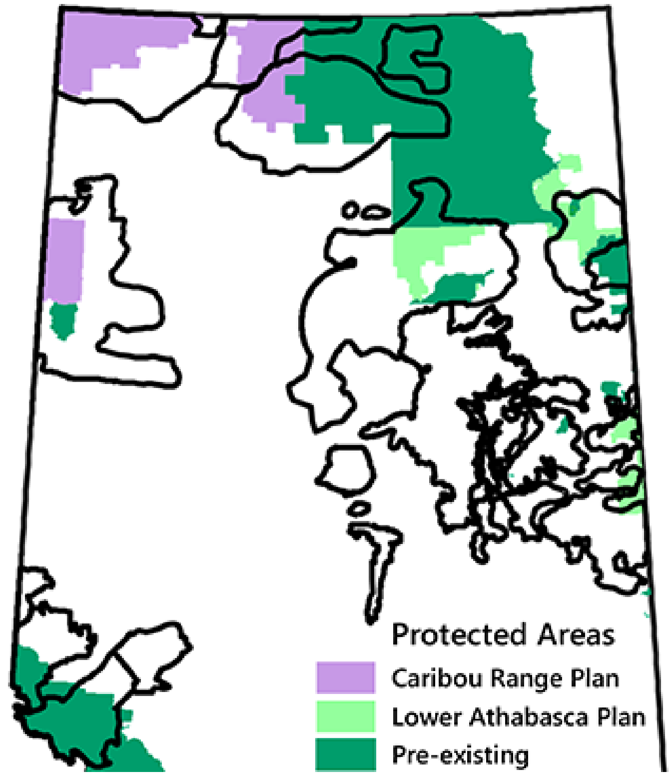


Fig. 11.17. The Alberta Caribou Range Plan proposed several new protected areas in northwest Alberta. Caribou ranges are outlined in black.

The planners did not anticipate opposition to the new protected areas. The resource potential of the proposed reserves was inconsequential when compared with most other parts of northern Alberta (Fig. 11.12c). No forest tenure was affected (by design) and the few existing oil and gas leases were to be grandfathered in (Denhoff 2016).

Local communities saw things differently. Grand provincial-scale trade-offs and relative resource values were not of interest to them. As they saw it, their prospects for growth were being unfairly constrained. A municipal committee in northwest Alberta was formed and it delivered a petition to the Alberta government containing over 9,000 signatures—approximately 50% of the regional adult population. They asked the government to "forgo any type of additional permanent conservation land designation" (NWSAR 2016). They also requested that a comprehensive socio-economic assessment be done.

The Alberta government found itself between the proverbial rock and a hard place. On one side were the vocal concerns of local communities, and on the other, the pressing demands of SARA and the public supporters of caribou conservation. The Alberta government's response was to send a letter to the federal government, in early 2018, indicating that additional time and an infusion of federal funds would be needed to advance caribou recovery in the province. The letter also laid out a clear challenge to the federal government: "Alberta's approach to protecting caribou populations and fulfilling the requirements under federal law cannot and will not come at the expense of our economy" (Phillips et al. 2018, p. 2).

As of this writing, the federal government has not indicated how far it will go in defending SARA. If it pushes too hard, Alberta will rebel. If it does not push hard enough, it will find itself in court for failing to uphold federal law. This is likely to be the major showdown over SARA that many have predicted since its inception in 2002.

Analysis and Conclusions

This case study features complex multi-tiered decision making. Wildlife managers knew decades ago that continued caribou declines were inevitable unless resource development was constrained. However, they had no authority over the decisions that mattered most, such as tenure allocations, harvest rates, and infrastructure planning. Stakeholder planning committees had no control over these decisions either, though this was perhaps moot since these committees were perpetually deadlocked.

So it was that the key decisions that determined the fate of caribou were not made deliberately by those responsible for managing caribou, but passively by other government departments pursuing the broader government agenda of economic development. Given the high profile of caribou, this cannot be considered an oversight. It was a conscious, if informal, decision by higher levels of government to prioritize economic development over caribou.

It was the federal government, through SARA, that ultimately forced Alberta to take substantive action to recover caribou. This is a good illustration of the difference between policy and law. The Alberta government had designated caribou as threatened in 1985 and had repeatedly committed to sustaining caribou in various policy documents. But there were no consequences when it failed to follow through on these commitments. In contrast, the directives of SARA carry the force of law and cannot be ignored, as the federal government has already learned through several successful court challenges.

Within the federal recovery strategy, the inclusion of a disturbance threshold was pivotal. This was one of the most important decisions made in the history of caribou conservation. For better or worse, it brought clarity to all parties about what caribou recovery would entail—it drew a line in the sand.

The choice of 65% undisturbed habitat as the management target was based on a blend of science and subjective judgment. The statistical relationship between habitat disturbance and caribou persistence provided an objective foundation for decision making. However, the selection of 65% as the breakpoint, corresponding to a 60% probability of persistence, was a subjective decision. It reflected the recovery team's assessment of acceptable risk and perhaps their assessment of political feasibility. Such decisions tend to be attacked from all sides (too risky, too impractical), yet without them, nothing happens.

Another dimension of decision making highlighted by this case study concerns the rights of local communities with respect to public lands. As illustrated by the response of northern Albertans to the range plan, even marginal resources that do not currently support anyone's livelihood can motivate forceful opposition to habitat protection by resource-dependent communities. Given their close relationship with the land, should these communities have a veto over conservation decisions? Or should SARA have the veto because it conveys the conservation goals of broader society? What about local Indigenous communities, which have their own perspectives and rights concerning land use? Clearly, some form of balance needs to be achieved, which leads us back to regional planning.

This case study also features an important ethical question: is it acceptable to kill one species (wolves) in order to save another (caribou)? Opinions are divided, even among conservationists. Certainly, the desire to forestall near-term extirpation is a compelling argument for taking extreme measures, including wolf control (Hervieux et al. 2014). However, opponents of wolf control argue that wolves are being killed mainly because of the government's unwillingness to address the root causes of caribou declines, which relate to ongoing habitat degradation (Proulx et al. 2017). There is merit to both perspectives and the final decision is likely to be made in the court of public opinion.

Notably absent from the planning processes to date has been a formal assessment of socio-economic trade-offs. Under SARA, this aspect of conservation decision making is meant to be addressed at the action planning stage. However, the Alberta range plan did not include a socio-economic assessment.

The absence of socio-economic considerations in the Alberta range plan does not mean they have been overridden or are no longer important. The province's strongly worded letter to the federal government makes it clear that caribou conservation will not come at the expense of the provincial economy. Moreover, the federal caribou recovery strategy makes allowances for economic contingencies: "Implementation of this strategy is subject to appropriations, priorities, and budgetary constraints of the participating jurisdictions and organizations" (EC 2012c, p. iv). Thus, planning remains incomplete—the economic cost of recovery is a shoe that has yet to drop.

We turn finally to an appraisal of what has been achieved through caribou recovery efforts to date. The short answer is: very little, other than research. The most significant achievement has been the development of the provincial caribou range plan which, for the first time, contains concrete management targets and a plan for achieving them. But in practical terms, the only reason caribou herds no longer face imminent extirpation in Alberta is that wolves are now being killed on a massive scale. Long-term recovery is still far from assured.

It remains to be seen if the range plan will be implemented as proposed. There are many technical challenges to be resolved and economic hurdles to be crossed. The federal government's involvement and the legal weight of SARA should ensure that an earnest effort is made. But it is unclear how far the federal government will intervene if Alberta backtracks because of costs.

There are also concerns with the plan itself. The chosen disturbance target provides only a 60% probability of stability, which is far from reassuring. Furthermore, it is unclear whether achieving the habitat target will produce the expected outcomes. Everything hinges on a statistical relationship between disturbance and caribou recruitment that was based on a small number of coarse-scale observations. The reliability of this relationship when applied to fine-scale planning efforts is unknown.

The plan also includes several forms of unacknowledged risk. The decision to allow continued industrial disturbances within caribou ranges while restoration gets underway—in apparent contravention of federal policy—prolongs the entire recovery process, exposing caribou to increased risk. In addition, the plan does not take the effects of climate change into account. This is a critical oversight, given the virtual certainty that herds will have to shift their ranges northward as temperatures warm (Dawe and Boutin 2016). Disturbance management and restoration efforts should anticipate these range shifts by targeting both current and future caribou range. Finally, the plan makes no provisions for the future impacts of fire.

Given these risk factors, the odds are low that all herds will persist. This being the case, perhaps triage would have been best after all. However, there is an important counterargument to be considered. In their analysis, the proponents of triage failed to account for the broader biodiversity benefits of caribou conservation.

The Alberta range plan is, at heart, a cumulative effects management system that will benefit many species. In a roundabout way, it achieves what the *Land-Use Framework* initially set out to accomplish, which was to manage the total industrial footprint through integrated planning. Whether 65% undisturbed habitat is the appropriate target is an open question. But it is undoubtedly a good starting point. The upshot is that the range plan does the right thing for perhaps the wrong reason.

Caribou seem to have provided the focus and political momentum needed for managing cumulative effects. Perhaps the coarse-filter approach is just too abstract for high-level political decision making. It may be that a concrete issue, like caribou viability, is needed. But there are also drawbacks to a single-species approach. Caribou, while wide-ranging, are not found everywhere. Moreover, they actively avoid many habitat types as a consequence of their predator avoidance strategy. The high vulnerability of many herds also raises an important question: what happens to conservation if a herd is extirpated? Similarly, what happens if a herd shifts its range because of climate change?

These concerns could be resolved by extending the caribou range plan's land management approach to the full working landscape. This is not inconceivable. If it can be shown that cumulative effects can be managed on caribou range without unacceptable economic repercussions, resistance to wider application may diminish.

Case Study 4: Swift Fox

Background

This case study examines the reintroduction and subsequent recovery of swift foxes in southern Alberta and Saskatchewan. The swift fox (Fig. 11.18) is a grassland species that once ranged across the Canadian Prairies and the US Great Plains (Pruss et al. 2008). Populations underwent precipitous declines in the late 1800s with the influx of Europeans to the West (Sovada et al. 2009). One of the main factors responsible for the decline was trapping. According to Hudson Bay records, an average of 5,000 pelts were harvested each year in Canada in the 1870s. In addition, large numbers of foxes were killed through poisoning programs targeting wolves.

Fig. 11.18. The swift fox was extirpated from the Canadian Prairies in the early 20th century and then reintroduced in the 1980s. Credit: L. Carbyn.

Swift foxes were also impacted by the ecological changes that accompanied the settling of the prairies (Herraro 2003; Sovada et al. 2009). With the disappearance of bison from the plains, scavenging opportunities for swift foxes were much reduced. The plains wolf also disappeared, leading to an increase in coyote populations which, in turn, became a major cause of swift fox mortality. Campaigns by ranchers and farmers to kill badgers and ground squirrels reduced the number of escape holes and denning sites. Finally, a large proportion of swift fox habitat was converted to cropland and pasture.

By 1900, reports of swift fox were rare in Canada and the northern US. The last Canadian sighting was made near Manyberries, Alberta, in 1938. In 1978, the swift fox was officially designated as extirpated in Canada. Populations in the US persisted.

Reintroduction Program

The reintroduction of swift foxes into Canada began with preliminary breeding efforts at the Alberta Game Park and the Calgary Zoo in the 1960s (Carbyn 1998). A more formal project got underway in 1972, when Miles and Beryl Smeeton established what is now known as the Cochrane Ecological Institute, near Calgary, and imported two pairs of swift foxes under permit from Colorado (Smeeton and Weagle 2000).

The Smeetons collaborated with Steven Herrero, at the University of Calgary, which led to a series of graduate student research studies. The initial studies examined the political and public acceptability of swift fox reintroduction and rated the suitability of possible reintroduction sites (Herrero et al. 1986). The question of whether it would be better to use imported wild-born foxes from the US for the reintroduction, rather than captive-reared animals, was studied as well. Attention was also given to release methodology, post-release monitoring, and funding sources.

The Canadian Wildlife Service became formally involved in the project in 1978, after COSEWIC designated the swift fox as extirpated in Canada. A few years later, the Alberta and Saskatchewan governments became formally involved as well, and the initiative transitioned to an inter-agency cooperative program. This brought additional funding and expertise and provided local jurisdictional oversight.

Fox releases began in 1983 and continued until 1997. The releases occurred within the core of the historical Canadian range, in two main areas: one centred on the border between Alberta and Saskatchewan, and the other in south-central Saskatchewan (Fig. 11.19). A total of 932 foxes were released. Of these, 841 were reared in the Smeeton facility—the descendants of 17 wild pairs from Colorado, Wyoming, and South Dakota (Smeeton and Weagle 2000). The remaining 91 releases were wild foxes obtained from Colorado and Wyoming.

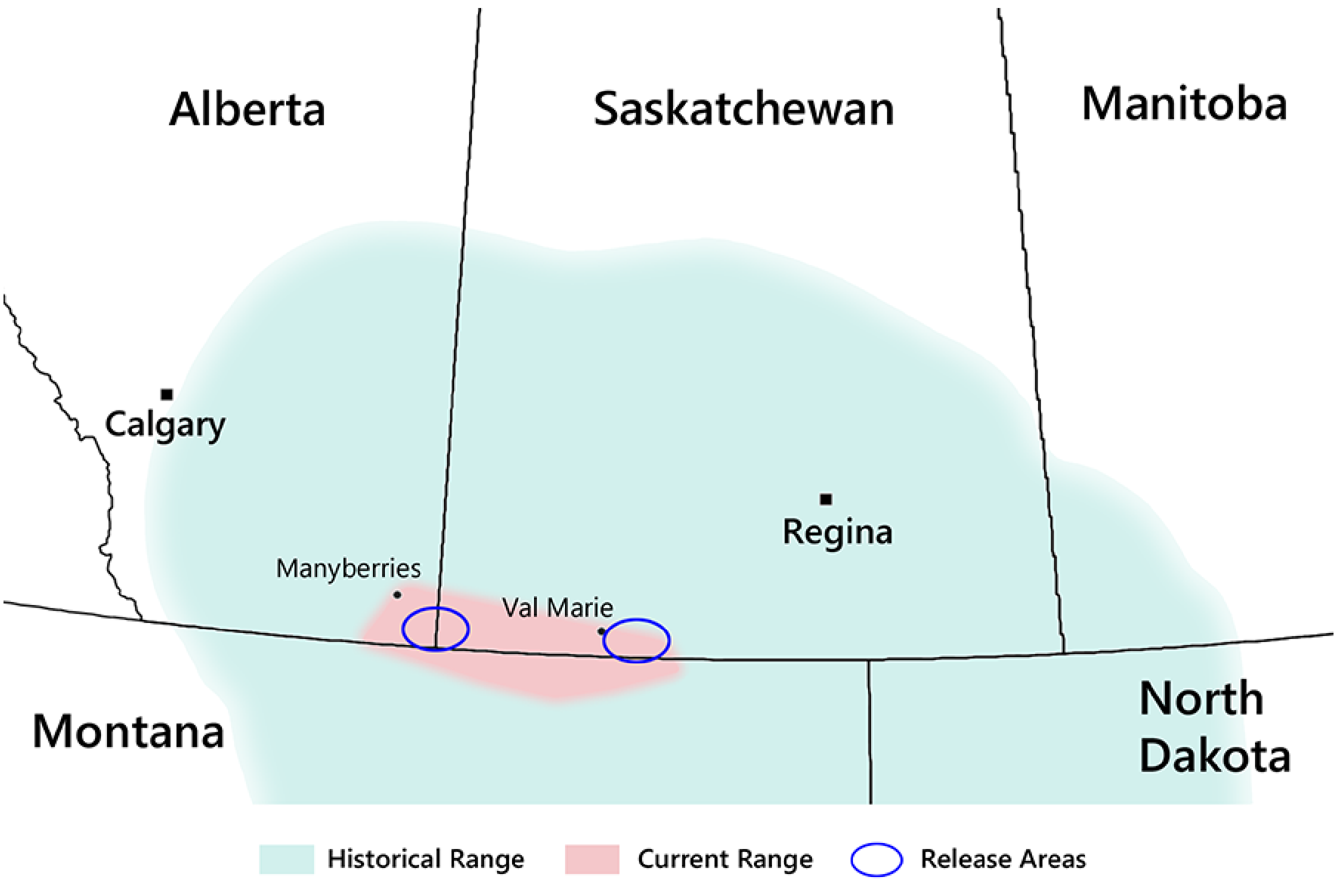


Fig. 11.19. The historical and current (2015) range of northern swift foxes. The two reintroduction areas are also shown. Adapted from: Smeeton and Weagle 2000 (release sites), Sovada et al. 2009 (historical range), and Moehrenschlager and Moehrenschlager 2018 (current range).

A second project began in 2004, with the aim of reintroducing swift foxes on the traditional territory of the Blood Tribe in southern Alberta (Pruss et al. 2008). Fifteen foxes were released in the first year, but the project was then discontinued because of a lack of funding.

A major concern for the reintroduction team was maintaining genetic diversity. Therefore, the source animals for the breeding operation were collected from several different US populations (Herrero et al. 1986). Breeding was carefully controlled to minimize inbreeding. Also, animals from the same bloodlines were not repeatedly reintroduced into the same geographic area (Smeeton and Weagle 2000). Outbreeding depression from the mixing of diverse genotypes was an acknowledged concern but did not seem to affect breeding success at the Smeeton facility (Herrero et al. 1986). There was also a concern that the source animals may lack the genetic adaptations needed to thrive in a Canadian climate. Since no local populations were available to use as source stock, this risk was accepted as one of the many uncontrollable factors inherent in the reintroduction.

Most aspects of the reintroduction program employed an adaptive management approach. This adaptive process did not involve quantitative modelling or formal experimental design. Rather, it was a classic example of "learning by doing" that involved trying different approaches and determining which was most effective. The outcomes were mainly assessed in terms of the rate of survival of the released foxes. The management levers amenable to study included (Waters 2010):

- Source of released foxes (captive-reared vs. wild)
- Location of release (considering variations in local terrain and vegetation)
- Timing of release (spring vs. fall)
- Age at release (juvenile vs. adult)
- Release method (soft, hard, intermediate)

The reintroduced fox population quickly grew and spread beyond the initial release areas, including into northern Montana. Thus, the reintroduction was considered a success. Swift foxes were downlisted from Extirpated to Endangered in 1999 and subsequently from Endangered to Threatened in 2012.

In the most recent field survey, in 2015, a population decline was observed instead of continued growth (Fig. 11.20). Managers have not been able to pinpoint the cause of this decline. The winter of 2010/2011 was particularly severe, and this may have been a contributing factor (Moehrenschlager and Moehrenschlager 2018). It is also possible that the population has reached a plateau. The region's carrying capacity for swift foxes is not known, but it is certainly much lower today than it was in the past (Herraro 2003).

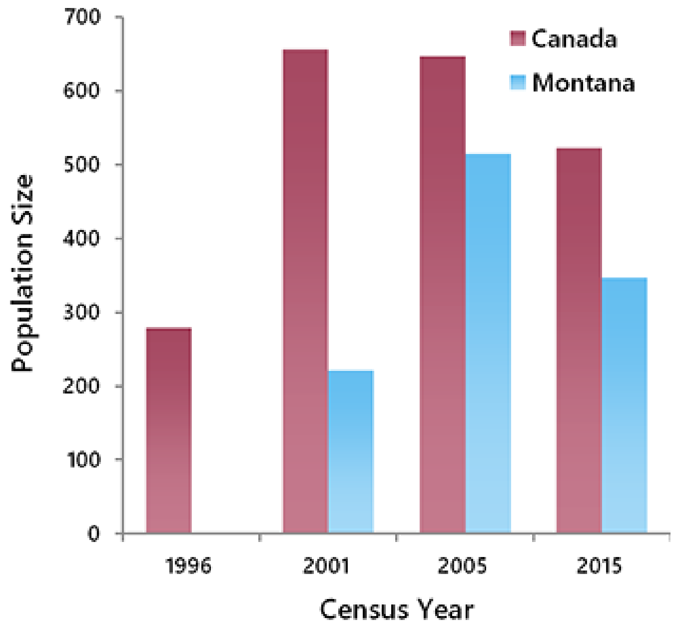


Fig. 11.20. The estimated change in size of the reintroduced swift fox population based on successive field surveys. Sources: Moehrenschlager and Moehrenschlager 2001; Moehrenschlager and Moehrenschlager 2018.

Multi-Species Action Planning

The federal swift fox recovery strategy was not completed until 2008, well after the reintroduction took place (Pruss et al. 2008). The stated recovery goal was to "restore a self-sustaining swift fox population of 1,000 or more mature, reproducing foxes that does not experience greater than a 30% population reduction in any 10-year period" (Pruss et al. 2008, p. 11). In terms of recovery actions, the strategy emphasized research, reflecting the limited state of knowledge about the species. The main threats to swift foxes were known, but the extent to which these threats would impair their recovery was unclear.

One of the research priorities was to identify critical habitat, an element missing from the recovery strategy. Other topics included the impacts of anthropogenic landscape disturbances and the biotic interrelationships between swift foxes, coyotes, and red foxes. The strategy also called for ongoing monitoring.

As far as actual management interventions, the strategy recommended that best practices for landowners be developed and disseminated through an outreach program. Swift fox recovery efforts were also to be integrated into a larger, unified conservation program for southern prairie species. This latter recommendation was realized in 2017, with the release of the *Action Plan for Multiple Species at Risk in Southwestern Saskatchewan: South of the Divide* (ECCC 2017b).

The multi-species action plan applied to nine species at risk and four species of special concern in southwestern Saskatchewan, including the swift fox. Compared with the swift fox recovery strategy, the multi-species action plan emphasized management action over research. The proposed activities included:

- Develop and implement grazing systems that provide high-quality habitat for species at risk
- Provide incentives to support targeted conversion of cropland and tame pasture to native grasses and shrubs
- Develop and encourage integrated pest management
- Develop an approach for infrastructure development that reduces disturbance and accidental mortality to

species at risk
- Manage fire in ways that benefit species at risk without threatening infrastructure and agricultural values
- Investigate the utility of conservation agreements to protect critical habitat
- Conduct outreach to help landowners reduce the environmental impacts of their activities

The multi-species action plan also partially identified critical habitat for all species at risk. For the swift fox, critical habitat was identified using a spatially-explicit habitat suitability model (ECCC 2017b). This statistical model linked swift fox habitat use, derived from a population survey in 2005, with 14 landscape-scale habitat variables, obtained from remote sensing data. The reliability of the model was verified by comparing its predictions against three separate population surveys that had not been used for model development. Critical habitat was defined as all areas within the planning region that were expected to support swift foxes based on modelling extrapolations. This amounted to approximately half of the current swift fox range in Saskatchewan. In contrast to caribou, population persistence was not taken into consideration, mainly because no population model was available.

A shortcoming of the multi-species action plan was that it gave little consideration to implementation—it was mainly a shopping list of potential actions. This deficiency was addressed in a subsequent planning exercise which used a structured decision-making approach to define implementation priorities (Martin et al. 2018; Carwardine et al. 2019). In this exercise, the various strategies listed in the multi-species action plan were treated as management alternatives to be assessed and compared. The goal was to identify the strategy, or combination of strategies, that maximized overall species recovery, taking the expected benefit, cost, and feasibility of each strategy into account.

In the decision analysis, the benefit of a given strategy was expressed as the probability of achieving the recovery objectives for each species (calculated for each species individually and then summed across all species). The cost of a strategy was the total cost of all individual actions as well as the value of foregone development opportunities. The overall feasibility was defined as the probability that the action would be implemented (given socio-political considerations) multiplied by the probability of success (given methodological and logistical limitations). All assessments were based on a 20-year time horizon.

The information required for the assessments was elicited from a group of experts using a structured Delphi approach during an intensive three-day workshop. The group was composed of nine grassland ecosystem experts from government agencies and universities. Many of these experts were contributors to the original multi-species action plan. To gauge uncertainty, the experts were asked to provide their best guess for each parameter as well as estimates of upper and lower bounds.

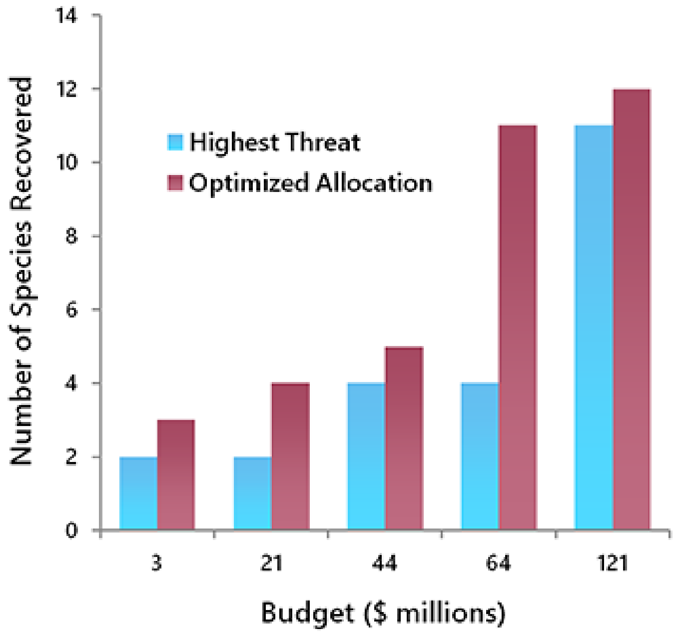


Fig. 11.21. The number of species at risk in southern Saskatchewan predicted to have a 50% or more probability of their achieving their recovery objectives, under a range of conservation budgets. The budgets were allocated using either optimized resource allocation methods (red) or by prioritizing the most threatened species (blue). There were 13 species in all. Adapted from Martin et al. 2018.

Once the required information had been assembled, the planners used linear programming to identify the optimal management approach. "Optimal" was defined as the strategy (or combination of strategies) that maximized the number of species that achieved a specified level of recovery success while minimizing the total cost. The team found that, for most budgets, optimal resource allocation provided better conservation outcomes than the common approach of prioritizing the most threatened species (Fig. 11.21). The size of the conservation budget had an overriding effect on the number of species recovered.

A limitation of the optimization exercise was that quantitative data were generally lacking; therefore, the assessments had to be based on expert opinion. Some of the estimates, particularly those regarding feasibility, were essentially informed guesses. Moreover, some management options were excluded from consideration altogether because of knowledge gaps. Thus, the results are best thought of as hypotheses that reflect the current state of knowledge.

Analysis and Conclusions

The reestablishment of swift foxes in Canada stands as a major accomplishment. Credit goes to the conservation practitioners whose dedication and determination made the reintroduction happen. Credit also goes to the swift fox, whose adaptability and high reproductive capacity allowed it to grow despite profound alterations of the prairie landscape (the point being that we cannot expect similar outcomes with all species at risk).

Fig. 11.22. A swift fox being released into the Saskatchewan prairie. Credit: L. Carbyn.

The dip in population size observed in the 2015 survey is a point of concern and suggests that the recovery of the swift fox is not yet assured. This decline may just be a weather-related fluctuation, but it is also possible that the carrying capacity of the current prairie ecosystem is relatively low. If the latter, then the future trajectory of swift fox recovery may depend on the extent to which the ecological integrity of the southern prairies can be restored. Saskatchewan's recent policy decision to privatize its rangelands (CWF 2017) will likely make this more difficult. On the other hand, the formal identification of critical habitat in the federal action plan means that habitat protection will now be a legal requirement.

This case study illustrates the convoluted path that conservation initiatives sometimes follow. At different stages, the reintroduction program was a private venture, a University of Calgary research project, an inter-agency cooperative initiative, and a component of the national species recovery program. A project like this takes shape when a few key individuals become convinced of its merit and provide the drive and determination to move it forward. These leaders attract others to the project, and the necessary resources are then collectively assembled. The lesson here is that conservation is sometimes a bottom-up process, dependent on individual initiative, rather than a top-down, policy-driven process.

Because the reintroduction of the swift fox posed no threat to farming or ranching operations, it encountered little opposition. Nor did the program have a high public or political profile. This was an example of the biocentric model of conservation in action, where scientific problem solving predominated. There was a single, clear objective—swift fox reestablishment—and decisions were largely data driven and relied heavily on structured learning.

After the reintroduction program ended, in 1997, the emphasis of management shifted to ensuring long-term persistence. However, there was uncertainty about what needed to be done. Moreover, there was a lack of clarity about what the recovery objectives should be, beyond basic population survival. Federal policy describing the full spectrum of recovery objectives was not provided until 2016 (ECCC 2020). As a result, the 2008 recovery strategy emphasized research and prescribed little management action.

As the recovery process evolved, the social dimension of conservation became increasingly important. Most of the proposed conservation measures in the 2017 multi-species action plan entailed changes to land-use practices, especially agriculture. This meant that the support and cooperation of land users was required, in contrast to the initial reintroduction phase.

The extension of recovery planning to multiple species, through the multi-species action plan, was an important step. It raised the profile of regional conservation needs, facilitated conservation synergies, and increased overall planning efficiency. In contrast to the 2008 recovery strategy, the action plan emphasized management actions over research. However, it still failed to meaningfully consider implementation. This gap was later addressed through an optimization exercise that identified management priorities, taking costs and feasibility into account. Together, the action plan, which identified strategies, and the optimization exercise, which identified priorities, provide a useful template for future recovery planning efforts.

When the optimization study was released in 2018, the headline it generated in the *Globe and Mail* newspaper was: "Too expensive to save? Why the best way to protect endangered species could mean letting some go" (Semeniuk 2018). The message the media latched onto, presumably because of its shock value, was that we must allow some species to become extirpated in order to ensure the viability of others—a classic triage interpretation. This reflects a misunderstanding of what the optimization initiative was meant to accomplish.

The aim of the optimization exercise was to identify management strategies that provided the greatest overall conservation gain within the study area, for any given conservation budget. The process did not pit one species against another. Instead, it was concerned with the effectiveness of individual management actions, giving preference to actions that benefited multiple species over actions that provided narrow benefits.

Notably, the optimization exercise did not try to determine how large the conservation budget should be (or, by extension, how many species had to be "let go"). The determination of what society is willing to pay for maintaining species at risk is not something that can be addressed by a group of technical experts at a workshop. Such decisions are inherently political because they entail finding an acceptable balance among competing social objectives.

The way that optimization exercises feed into the broader political dimension of conservation is by drawing attention to conservation needs. A graph like Fig. 11.21 vividly illustrates the conservation consequences of inadequate conservation funding, making trade-offs explicit. This provides a useful foundation for public dialog about land use. Moreover, by framing the debate in terms of the collective needs of a large suite of species at risk, the case for conservation is considerably enhanced.

In summary, optimal resource allocation should be used to ensure that existing conservation resources are used as effectively as possible. It is also vital to expand the resources that are available—it is not just how we slice the pie, but the size of the pie that is important. This requires public outreach to raise awareness of conservation issues and to draw attention to funding shortfalls. It also requires engagement at higher levels of the decision hierarchy, where budgets are determined.

Case Study 5: Walleye

Background

This case study examines the collapse and recovery of walleye from Alberta lakes (Fig. 11.23). The account is provided by Dr. Michael Sullivan, a fisheries scientist with the Alberta government who played a lead role in the recovery program.

A combination of factors makes Alberta's lakes highly susceptible to overfishing (Sullivan 2003). Whereas many provinces have thousands of lakes, Alberta has only about 800 lakes that support sport fishing, and of these, only 177 contain walleye. Furthermore, fish grow slowly and mature late in Alberta because of the cold northern climate. Finally, walleye are more easily caught in Alberta than in northern US lakes because prey availability is low (Mogensen et al. 2013).

Fig. 11.23. Alberta walleye populations crashed in the twentieth century as a result of overfishing. Credit: E. Engbretson.

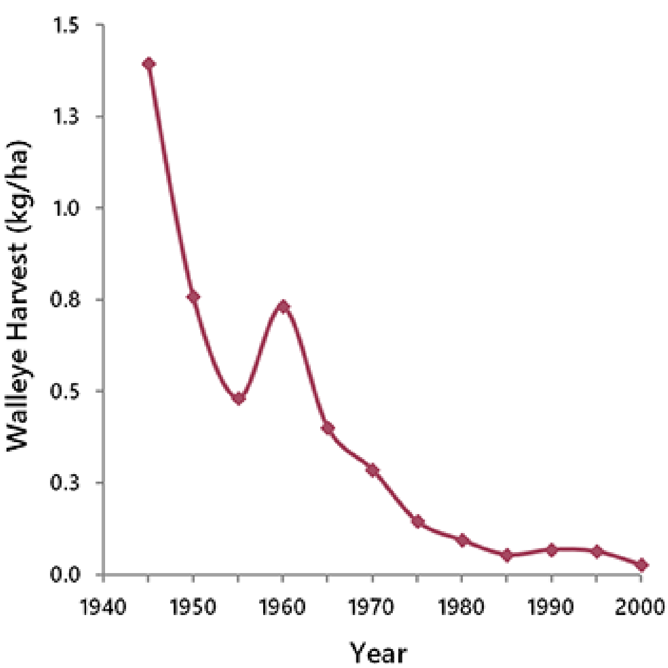


Fig. 11.24. Commercial harvests of walleye from six lakes (La Biche, Calling, Touchwood, Wolf, Beaver, and Moose) in Alberta during 1940–2000. Source: Alberta Fish and Wildlife Division records.

Alberta's largest lakes were fished heavily for subsistence and dog food during the fur trade period, especially in the late 1800s. This was followed by intensive commercial fishing, which peaked in the early twentieth century. By the late twentieth century, walleye were nearly extirpated from Alberta's large lakes (Fig. 11.24).

Smaller lakes were initially spared because they were less attractive to large commercial operators. These lakes later came under increasing pressure from sport fishing, as Alberta's population, and the number of anglers, increased rapidly in the second half of the twentieth century (Pybus and Rowell 2005). Sales of angling licenses climbed from 122,000 in 1965 to 343,000 in 1986 (Sullivan 2003). Furthermore, the industrialization of Alberta's north made most lakes in the province easily accessible. The effect of having a large number of anglers concentrated on a relatively small number of lakes was, in hindsight, predictable. By the 1980s, individual lakes with popular sport fisheries showed severe declines in sport fish populations, and the size and age structure of these populations were highly disrupted (Sullivan 2003). The loss of top predators led in turn to ecosystem disruption, through an explosion of prey species.

During the period of general collapse, many biologists in the provincial fisheries department did not realize or accept that a decline was occurring. Walleye populations in larger lakes had been collapsed for decades, and many biologists accepted this as the norm. Moreover, there was a strong and widespread belief among many biologists and anglers that sport angling had no detrimental effects. Reports of poor fishing in smaller lakes were attributed to a variety of causes, such as a decline in angler skills ("too many city anglers going to these lakes"), a sense that memories from older times were false, changing fish behaviour, and a belief that environmental conditions had changed. The lower catch rates measured in creel surveys were attributed to poor study design. As a result, lakes with walleye populations near extirpation were kept open to harvest.

By the late 1980s, anglers at traditionally good walleye fisheries were catching very little. Success rates were typically under 10% (i.e., 90% of anglers caught nothing). Angling licence sales were falling, and public complaints were rising. Some biologists, applying the then-new technique of computer modelling, proposed experimental harvest regulations involving fish size limits. These proposals and methods were met with considerable doubt and suspicion from senior biologists and fisheries bureaucrats. The outcome was more studies, but little action.

Recovery Planning

In the 1990s, stakeholders became more directly involved in walleye management, and there was also a new emphasis on evidence-based decision making. A key step was the construction of an age-structured population dynamics model, which synthesized knowledge gained through years of field studies (Fig. 11.25). Mortality was overwhelmingly caused by human harvesting, and this was the focal point of management interest. Four mortality variables were included in the model: the number of anglers, the mean harvest per angler, the age and size of fish taken, and the level of compliance with fishing regulations. The reproductive capacity of walleye was incorporated into the model as a set of fixed parameters. The model also included a set of management options linked to these core variables. Testing and refinement of the model were undertaken to ensure consistency with observed patterns.

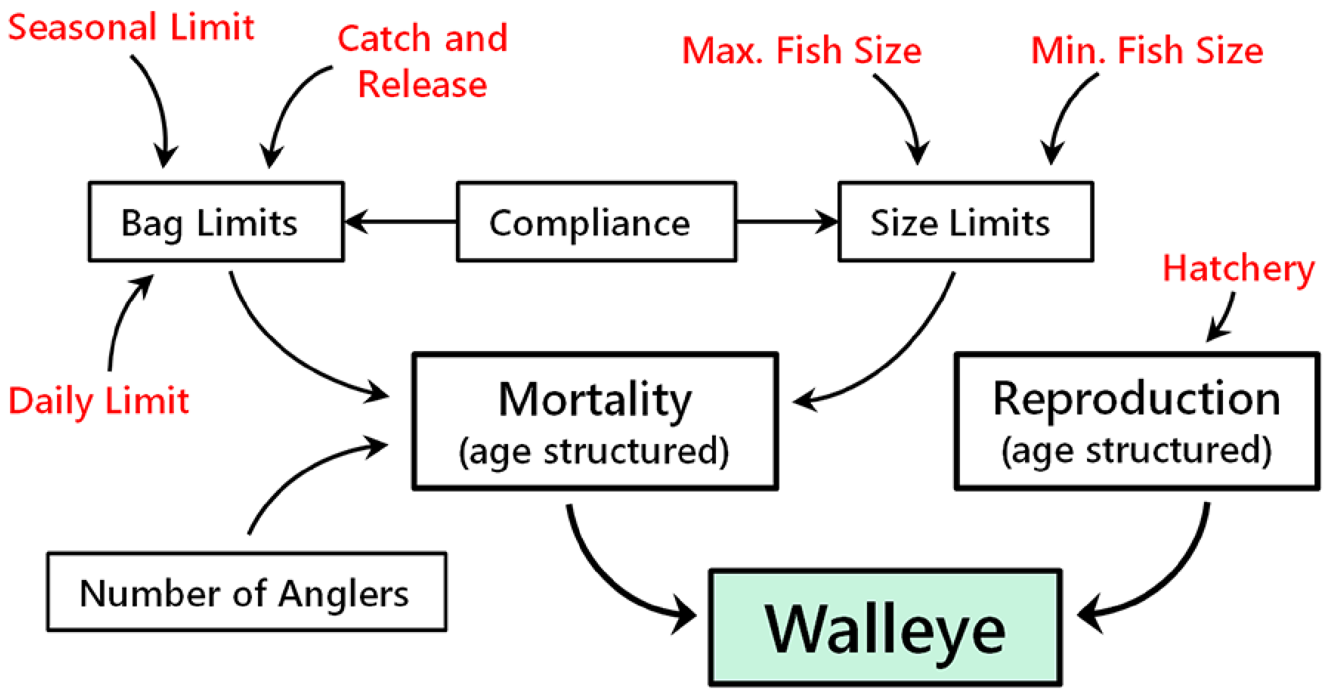


Fig. 11.25. The conceptual model of walleye population dynamics used to support recovery planning. The items in red represent management levers that were explored in stakeholder workshops.

Through their explorations, the biologists leading the modelling effort developed an understanding of the management changes needed to achieve walleye recovery. But they also knew that management by decree would not be successful. The imposition of new constraints on anglers was bound to set off protests and calls to the responsible minister. And while attitudes within the fisheries agency had begun to shift, senior management was still not convinced that the proposed management approach was appropriate.

To gain the support needed for implementing new recovery measures, the biologists held a series of stakeholder workshops in 1995 and 1996. The aim was to engage stakeholders in the decision-making process to help them understand the management options and the likely outcomes. It was hoped that a management approach could be identified that would have broad agreement among stakeholders and would be acceptable to senior management.

The biologists knew that careful selection of participants and structured facilitation of the workshops would be critical to making progress. Michael Sullivan describes the process that was used:

> We needed to have groups of concerned and open-minded people attend the workshops. At meetings open to the general public, it's common to attract a few highly vocal and opinionated individuals that dominate the discussion. This can make it difficult for everyone to be heard. Achieving openness while still ensuring fairness to all perspectives is a major challenge.
>
> Our solution was to limit the workshops to approximately 15 people each. That seemed to be a number that was large enough to obtain diverse opinions but small enough to have meaningful conversations. We

chose 20 meetings sites across the province, to allow the Minister to say that the process was not rural focused, or city focused, or aligned with special interests.

The individuals invited to each meeting were selected by local fisheries staff who were provided with a list of the types of stakeholders we were looking for. We asked for several sport anglers, several commercial operators, a local science teacher, a tackle store owner, a few retired people, a few young people, Indigenous people, the local Forestry official, Parks staff, and so on. We were looking for a range of perspectives. We wanted people who cared about the topic, but not just as anglers. We wanted people who might typically complain about the government and make them part of the process. We wanted people who would provide new viewpoints. We wanted diversity in knowledge, experience, opinions, and concerns. What we didn't want was an echo chamber. Every consultation exercise needs to wrestle with this list.

Once the list of potential names had been assembled, invitations to participate in the workshop were sent out. We offered to pay their expenses (gas, hotel, meals). We showed, with real sincerity, that their participation was important to us and to Alberta's fisheries. We treated them with respect, and as well-treated and good people do, they responded in kind.

Each workshop started with dinner. People are less likely to argue with someone they have just shared a meal with. After coffee and dessert, a presentation on the state of Alberta's fisheries was given and then a game was played to help the audience understand the process of modelling. In this loud and fun game, which simulated a fishery, people took turns drawing paper "fish" out of a cardboard box until none were left.

Each person drew a given number of fish based on realistic Alberta catch data. The size and fecundity of the paper fish also followed actual Alberta fisheries data. We then showed what it would take to maintain the stock, using different limits on harvest size and quantity. The point of this simple model was to get people to appreciate the cumulative effect of having many anglers catching a few fish. We also wanted them to understand the limitations of size limits, the minor influence of bag limits, and the overall complexity of management.

After the cardboard box game, we introduced the more complex computer model, which was run interactively and in real time. The input variables were displayed as "sliders" on a computer screen projected at the front of the room. The output was a time-series graph showing the predicted trajectory of the walleye population.

The initial runs were identical to the earlier game. Participants would soon say, "Heck, we already knew *that* bag limit wouldn't do anything." Then, we explored what might happen with a complex regulation. For example, we might ask if a 50 cm size limit, with a bag limit of two fish, non-compliance of 20%, and a 30% increase in fishing pressure would achieve recovery. Soon, the participants wanted to try their own complex combinations. The key to the process was in allowing the participants to accept the model on their own terms and incorporate its output into their own decision making. The playing-with-regulations game often went on for hours.

At the workshops, stakeholders invariably had questions that extended beyond fishing regulations. What about shoreline habitat destruction? Are you going to kill cormorants? Why not just stock more fish? These questions had to be answered meaningfully. Yet, we also needed to keep the workshop focused on the main drivers of walleye population dynamics, which were related to fishing. Our approach was to prepare for such questions ahead of the meeting. When hot-topic issues arose at the workshops, we generally had variables in the model, supported by field data, that allowed participants to explore them and then put them to rest.

At the end of the series of walleye workshops, the vast majority of participants had selected regulations that were based on good science, were simple to follow and enforce, and promoted fisheries recovery in a reasonable length of time. Approximately 80% of walleye lakes were recommended for catch-and-release fishing until recovered; 10% were classified as vulnerable, and only large fish could be kept; and 10% were classified as stable and received a broader harvest-size limit.

The Minister responsible for fisheries whole-heartedly accepted the workshop recommendations and they were implemented in 1996. Within a few years, signs of recovery were obvious. Catch rates climbed and the sizes and ages of walleye increased (Fig. 11.26). Eventually, fishing became better than most anglers, or even their grandparents, ever remembered. Now, in 2018, fishing for walleye within an hour's drive of Alberta's largest cities is better than it was in remote fly-in lakes in the 1980s and 1990s.

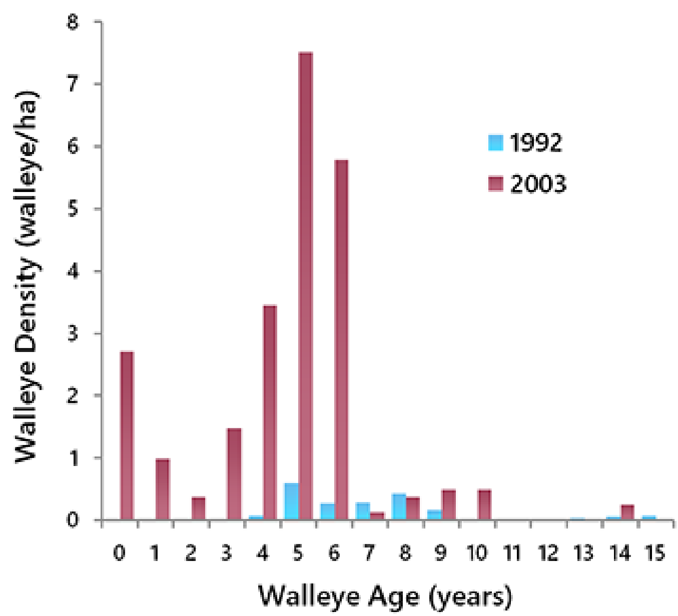


Fig. 11.26. The density of walleye at Wolf Lake, Alberta, by age class. The increase in density between 1992 and 2003 is attributed to the implementation of catch and release fishing regulations in 1996. Source: Alberta Fish and Wildlife Division records.

Analysis and Conclusions

In this case study, the various components of the decision-making process were tackled by different groups of people at different points in time. Like the Al-Pac case study, the elements of structured decision making were all present, but the process stretched out over many years.

The critical step of defining objectives evolved slowly and was linked to changing attitudes within the provincial fisheries agency. Senior biologists had come to accept low walleye populations as the norm, and from their perspective, there was no pressing problem that needed to be solved. This was an example of the shifting baseline syndrome in action (Pauly 1995). Agency culture was also a factor. Information was filtered and judgments were coloured by long-standing institutional norms. Change finally occurred when a new cohort of biologists brought fresh ideas, perspectives, and methods to the agency. With support from stakeholders, this group eventually prevailed, and walleye recovery, rather than maintaining the status quo, became the focus of management.

The decision-making process was divided into two distinct phases. The first phase was science based and was handled internally by a core group of biologists. It emphasized quantitative research and the development of a walleye population model. This phase provided the information and structure used in later stages of the decision-making process. In effect, the computer model served as a digital version of the consequence table described in Chapter 10.

The second phase emphasized social decision making and centred on a series of stakeholder workshops. At these workshops, little time was spent discussing the objectives, identifying management levers, or searching for the best available information. All of these aspects were directly imported from the first phase of the decision-making process. The stakeholder groups focused on exploring the outcomes of management alternatives and choosing the best option.

The robust approach to decision-making used in this case study was a major contributor to the success that was achieved. The development of a solid scientific foundation was vital, in that it enabled evidence-based decision making. Furthermore, planning efforts were not limited to the identification of threats and the listing of management options—a shortcoming of many recovery plans. Instead, the process was carried through to implementation, and considerable effort was devoted to the social dimension of decision making. This was vital for gaining a broad base of support for management changes. The biologists leading the process recognized that much would depend on trust, which had to be earned, and that the personalities of the participants would make a big difference.

Several intrinsic features of the management problem also contributed to the successful outcome. The walleye system was relatively simple and self-contained, with few confounding factors. Moreover, the main threat—overfishing—fell within the purview of the biologists responsible for managing the species. Finally, and perhaps most importantly, the long-term interests of anglers and walleye were fundamentally aligned. The more fish, the better the fishing experience. The main trade-offs were temporal. Anglers had to forgo some fishing opportunities today to benefit from greater opportunities in the future.

A later effort to apply the same planning techniques at the regional scale, for the recovery of several species at risk in the eastern slopes of the Rocky Mountains, met with less success (GOA 2017b). The process again began with scientific study and the development of population models. In this case, there were additional drivers in the system, besides fishing, that had to be incorporated, such as industrial development and invasive species. This meant that decision making was more complex and involved more stakeholders. It also meant that fisheries biologists no longer had authority over all the relevant management levers.

In the end, the process was derailed by opposition from the Alberta Fish and Game Association (AFGA) and its supporters. Though the fisheries biologists had conducted extensive outreach and held many workshops in support of the initiative, AFGA was resolutely opposed. AFGA's position was that anglers should not have to bear the cost, through reduced fishing opportunities, of habitat degradation arising from industrial development (AFGA 2018):

> Managers have not used all the tools available to them, particularly those that deal with habitat. … The Alberta Fish and Game Association cannot support the direction envisioned which relies primarily on angling regulations. … The plan does not appear to apply to industry, agriculture, infrastructure or urbanization other than assessing their impact on habitat. (pp. 3–4)

In support of its position, AFGA conducted its own outreach, mainly to its membership, and directed a lobbying effort targeting elected officials. The fisheries biologists countered with their successful track record in recovering walleye and assurances that the other species could be also recovered using the proposed measures. They also conducted opinion surveys that demonstrated strong public support for their earlier recovery-related efforts (GOA 2018b). Nevertheless, once the issue became politicized, the Minister balked. Instead of implementing the proposed recovery plan for the eastern slopes region in 2018, as scheduled, the Minister halted the process, stating that she "wasn't convinced the science was strong enough" (Rieger 2018), even though trade-offs among conflicting objectives were the fundamental issue and not the science.

We see here the political power that organized groups can wield (i.e., the "squeaky wheel" effect). A more general lesson is that the scale of planning is an important factor in determining conservation outcomes. In this case study, shifting from individual lakes to a broad region brought more stakeholders, more conflicting objectives, and a higher political profile. Arguably, it was this added complexity that was the fundamental barrier to success.

Case Study 6: Reserve Design

Background

This case study examines a planning process intended to guide the establishment of new protected areas in northwestern Alberta. This example illustrates the mechanics of systematic conservation planning, using Marxan, and provides insight into the handling of trade-offs among conservation objectives.

The initiative began in early 2015, motivated by the Alberta government's stated intention to extend regional planning to northwestern Alberta later that year. It was expected that the new regional plans would include a protected area component, following the precedent set by the *Lower Athabasca Regional Plan* in 2012 (see Case Study 2). Sensing an opportunity and a need, a group of conservation practitioners decided to conduct a study of regional conservation priorities, as a preparatory step (Schneider and Pendlebury 2016).

The government had not set a target for the amount of protection in the northwest, so the study was designed to be flexible with respect to targets. The group assumed the precedent set by the *Lower Athabasca Regional Plan* (i.e., 21%) was unlikely to be exceeded and that a lower amount was quite possible. The objective of the study was to determine where new reserves should be located to obtain the maximum conservation benefit, whatever the limit on the amount of protection might be.

There were two main tasks. First, the group had to select the biological entities to be protected. Second, given constraints on the amount of protection possible, the group had to develop efficient reserve designs for representing the selected biological elements.

A technical working group was established to conduct the study. It was composed of conservation scientists from the University of Alberta and representatives from most of the major conservation organizations operating in Alberta. The aim was to have a broad base of conservation expertise to draw on. In addition, the working group was meant to serve as a forum for resolving differences among organizations, so that the conservation community could speak as a single voice when providing recommendations to regional planners.

The working group included a Government of Alberta liaison linked to the regional planning process. Additional representatives from Alberta Fish and Wildlife, Alberta Parks, and the Alberta Biodiversity Monitoring Institute provided information and feedback. The group's efforts were supported by a grant from Alberta Ecotrust and in-kind contributions from group members and their organizations.

Selection Criteria

The group decided to combine the province's three northwest planning regions into a single large study area (Fig. 11.27). They felt this would provide added flexibility for achieving representation targets and result in more efficient reserve designs.

The group's first step was to review earlier conservation planning efforts. Of particular relevance was the example provided by the *Lower Athabasca Regional Plan* (GOA 2012). The group assumed that alignment with this plan would be necessary for their recommendations to be seriously considered by regional planners. The *Lower Athabasca Regional Plan* identified four criteria for the selection of sites:

- Areas that are representative of the biological and landform diversity of the planning region
- Areas with little to no industrial activity
- Areas that are roughly 4,000–5,000 km^2 in size (for maintaining ecological processes)
- Areas that support Indigenous traditional uses

The group also reviewed a planning framework developed by Alberta Parks in 2014 (AP 2014). This framework employed a hierarchical coarse-filter approach. It also had a complementary fine-filter component which focused on unique geologic features, rare and localized species, and critical habitat for wide-ranging species. This framework provided a methodology but did not identify preferred sites.

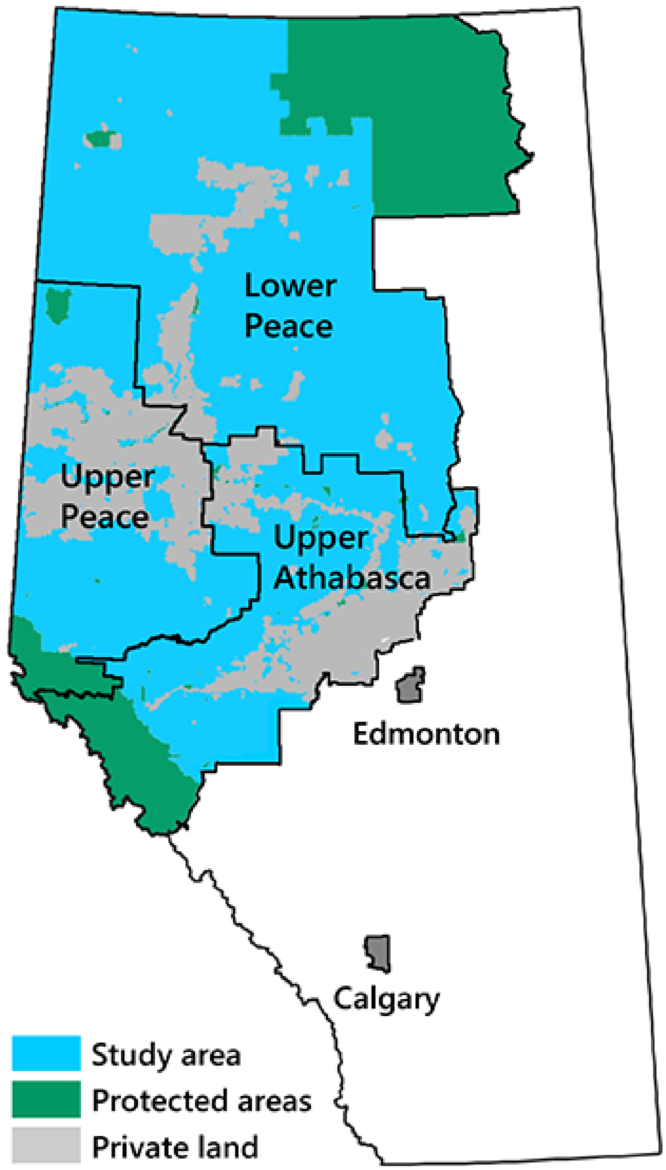


Fig. 11.27. The planning area for the reserve design study was composed of three land-use planning regions in northwestern Alberta, with a total size of 350,000 km^2. Private land (grey) was excluded from the analysis.

Finally, there were several map products that had been developed to support conservation planning in Alberta. These included a map of Environmentally Significant Areas, commissioned by the Alberta government (FBC 2014), and three maps of conservation priorities developed independently by three conservation groups. Each organization used a different scoring system for assigning conservation priorities.

The working group used a combined coarse- and fine-filter approach for its study, consistent with the principles of systematic conservation planning and the Alberta Parks framework. The choice of specific elements reflected the group's judgment about what was needed to achieve comprehensive representation of biodiversity. The choices were informed by earlier planning efforts and constrained by data availability. The coarse-filter elements included Natural Subregions (Fig. 11.28), major vegetation types, major wetland types, and a land facet layer developed by Dr. Scott Nielsen at the University of Alberta (Michalak et al. 2018).

The fine-filter elements included the habitat of all species of conservation concern for which spatial data were available (n=45). The habitat datasets were obtained from the Alberta Biodiversity Monitoring Program in the form of modelled species distribution maps. The fine-filter elements also included 19 fine-scale wetland classes and the modelled distribution of high-density waterfowl areas, provided by Ducks Unlimited Canada.

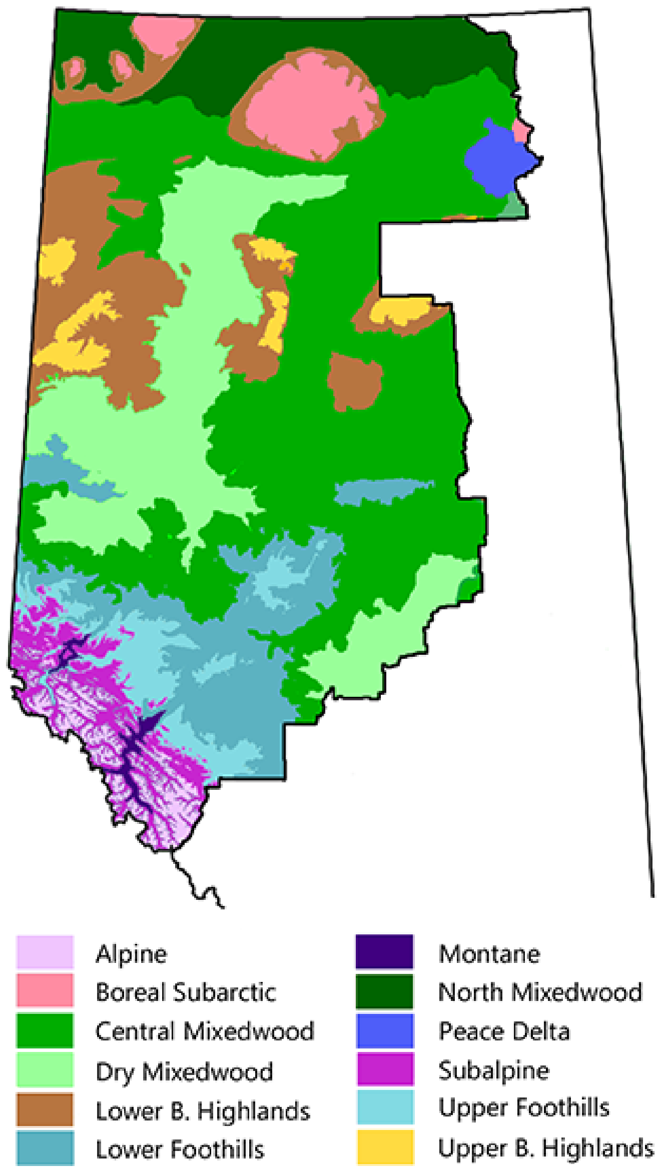


Fig. 11.28. The Natural Subregions of Alberta (NRC 2006) clipped to the study area.

Identifying Priority Areas

The working group used Marxan to identify potential reserve configurations (see Chapter 8). A variety of design scenarios were examined to reveal trade-offs among design objectives. Existing protected areas were automatically included in all scenarios. Privately owned lands (Fig. 11.27) were excluded from consideration in accordance with the regional planning process. Because Marxan typically finds many different reserve configurations that work equally well, the model was run 100 times for each scenario. The probability of selection was then calculated for each 500-ha planning unit.

A key challenge was that the government had not yet defined a protection target for the region. The group's solution was to generate reserve designs across a range of representation targets, from 5–30% of each coarse- and fine-filter element (Fig. 11.29). This way, relevant guidance would be available to regional planners regardless of the overall amount of protection they ultimately decided on.

In the base scenario, Marxan was required to achieve all coarse-filter representation targets while also prioritizing planning units with a low intensity of disturbance and minimizing the total area protected. The proxy for disturbance intensity was the density of linear features, obtained from the Alberta Biodiversity Monitoring Institute. A penalty was applied to the optimization algorithm to encourage the model to select contiguous patches rather than isolated planning units. The base scenario was first run using a 5% representation target for all coarse-filter elements. The process was then repeated with higher targets (i.e., 10%, 20%, and 30%).

The other scenarios were extensions of the base scenario. To begin, the group ran a scenario that included the fine-filter elements, with representation targets matched to the coarse-filter elements. The inclusion of fine-filter elements had no appreciable effect on the designs. This was recognized to be an artifact of data availability. The fine-filter species included in the study were those that the Alberta Biodiversity Monitoring Program had been able to survey. These species generally had broad distributions and were easily represented by the coarse-filter designs. Spatial data for true fine-filter species—those with unique and restricted habitats—were generally unavailable, and so these species were not included. This was acknowledged as a limitation of the study.

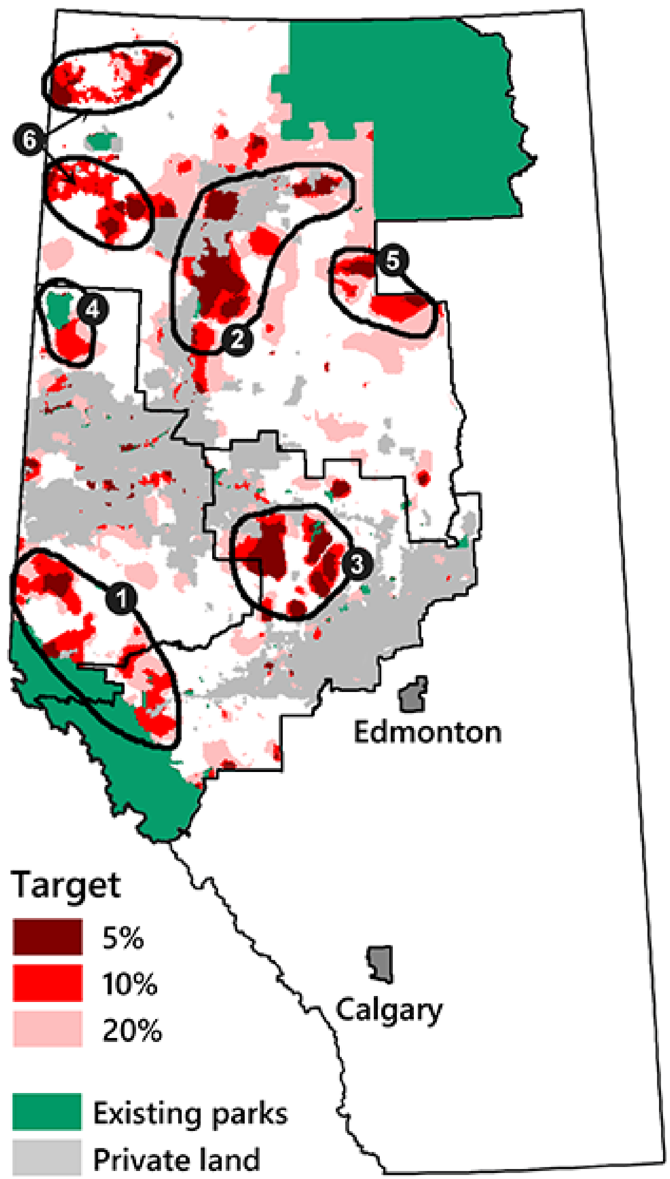


Fig. 11.29. A map of the planning units consistently selected in the base scenario with fine-filter elements added. Results are shown for representation targets set at 5%, 10%, and 20%. The six numbered areas outlined in black were identified by the working group as priority areas for establishing large, contiguous reserves (in rough order of priority). The two areas labelled 6 contain similar features and are interchangeable.

The group also explored a scenario that provided enhanced protection for a suite of high-profile species at risk residing in the foothills region east of the Rocky Mountains (i.e., woodland caribou, grizzly bear, Athabasca rainbow trout, bull trout, and Arctic grayling). In this scenario, planning units that contained habitat for three or more of these species were forced into the designs. The analysis showed that this could be done without increasing the overall area of protection. This was because the foothills region was already emphasized in the base scenario as a result of its low level of existing protection (just 1.4%). Prioritizing the species at risk in the foothills required only an adjustment in which specific parts of the foothills would be protected.

In another scenario, the group explored options for achieving the 65% target for caribou habitat protection prescribed by the federal recovery strategy. Preference was given to caribou habitat with the least disturbance intensity and the highest stability under climate change. In this case, there was a clear trade-off. The areas that were best for caribou did not align well with the areas that were best for representing biodiversity overall (Fig. 11.30). The implication was that, in the face of constraints on the total amount of protection, a choice would have to be made between protecting caribou and protecting overall biodiversity.

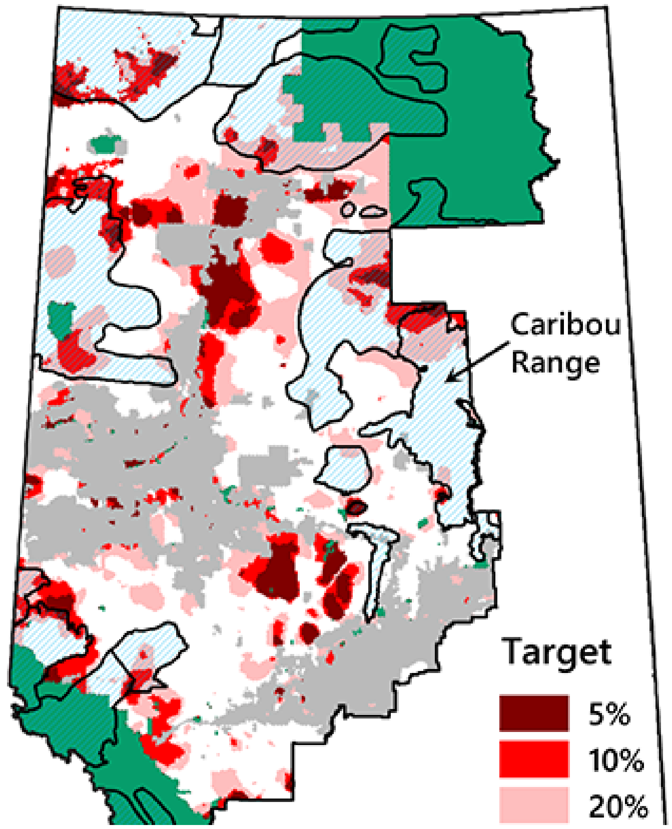


Fig. 11.30. A map overlay illustrating the limited overlap between caribou ranges (light blue) and the planning units selected in the base scenario with fine-filter elements added.

Lastly, the group examined a scenario in which Marxan was required to minimize conflicts with resource development while still achieving representation targets. The proxy for resource conflict was the estimated value of oil, gas, and forest resources within each planning unit (Map C in Fig. 11.12). The reserve designs for this scenario were not appreciably different from the base case. This was because resource values and linear disturbances were highly correlated spatially—most of the disturbance footprint was related to access for resource development. By requiring Marxan to avoid linear disturbances in the base model, it avoided areas with high resource potential by default.

Once the scenario analysis was completed, the working group turned its attention to making recommendations. The group recognized that regional planners would need more than just raw Marxan output to work with. A bridge had to be built between the technical analysis and its application in land-use decision making.

The group first developed a composite map of Marxan output for the base scenario with fine-filter elements added. This map illustrated the planning units consistently selected under the different representation targets. Next, the group identified six priority areas for protection (black ovals in Fig. 11.29). This step was needed because the Marxan designs were too fragmented to be used directly in regional planning. The intent was to identify the best sites for establishing large contiguous protected areas, as required to maximize ecological function and to align with the criteria established by the *Lower Athabasca Regional Plan*.

The group also provided a written description for each of the six priority areas, conveying their individual contributions to the overall system. The group felt that planners and stakeholders needed to understand why the specific sites were necessary and why they were not substitutable. The group also summarized the key findings from their scenario analyses and discussed the limitations of their study.

The group's final report was released in 2016, 15 months after the study began (Schneider and Pendlebury 2016). The Marxan data files were provided to the government through the government liaison in the working group.

Analysis and Conclusions

This case study provides an example of biocentric reserve planning. The working group had neither the capacity nor the authority to conduct a more comprehensive planning effort involving industry, local communities, and other stakeholders. Social decision making was expected to happen later, through the regional planning process. The group saw its role as preparing a sound foundation for later negotiations by identifying biotic priorities for protection.

This case study illustrates that there is a subjective aspect to setting protection priorities, even within the domain of conservation. Prior to this study, the government and several conservation organizations had prepared assessments of conservation priorities in northern Alberta, and each was distinct. The priority assessments were strongly influenced by the choices each organization made about which aspects of biodiversity to emphasize in its analysis.

A major contribution of this planning initiative was the opportunity it provided for alignment within the conservation community. Coming into the process, individual participants had different ideas about what was important to protect: species at risk, waterfowl and wetlands, old-growth birds, species richness, areas of high endemism, and so forth. As the list of elements expanded, it became apparent to the group that *all* habitats were important and merited protection.

The desire for comprehensive representation changed the nature of the planning exercise relative to earlier prioritization efforts. It made the coarse-filter approach the obvious foundation for the analysis. The coarse-filter approach was also recognized to be the most robust under climate change. The other major difference from earlier efforts was that this exercise incorporated constraints on the amount of protection. This motivated a search for optimal design solutions using Marxan. In addition, greater consideration was given to trade-offs among representation objectives.

From an ecological perspective, the final design was a compromise solution, as would be expected in the face of area constraints. It provided balanced representation of all elements, but it did not ensure that the critical habitat of all species was protected or that recovery objectives would be achieved. Woodland caribou are a prime example. The base scenario provided some protection of caribou ranges (mostly in existing reserves), but it was far below the level prescribed by the federal species recovery strategy. Shortfalls were also likely in other species, though this was not quantified because critical habitat was not identified for most.

The final design also suffered from a low level of connectivity. Given such a large study area, the priority regions for protection were separated by great distances (Fig. 11.29). The group identified some local opportunities for connectivity in their report, but no solutions were provided for resolving the major disjunctions. Achieving both representation and connectivity would require much greater levels of protection than considered in the study. Alternatively, connectivity could be achieved through special management zones.

These deficiencies need to seen in context. Designing an ecologically optimum reserve system was never the intent of the planning exercise. The aim was to identify conservation priorities, so that any new reserves arising from the regional planning process would be located in areas that provided the greatest conservation benefit. Constructing an ecologically optimum reserve system would require a level of protection substantially greater than the 21% precedent set in the *Lower Athabasca Regional Plan*.

In the end, the project failed to achieve its goal of guiding protected area planning in northwest Alberta. Even though the working group did everything in its power to align its efforts with the regional planning process, including having a government liaison participate as a group member, its findings never used. Instead, shortly after the group submitted its report, the provincial government announced its intention to protect the range of three caribou herds in northwest Alberta, as part of its caribou recovery efforts (see Case Study 3; Denhoff 2016).

Given limits on the amount of protection that is politically feasible, a decision to focus on protecting caribou leaves less opportunity for protecting sites that benefit biodiversity overall. The three new caribou areas are large, totalling over 15,000 km^2, yet they are largely devoid of the high-priority planning units identified in the Marxan analysis (Fig. 11.31). This is exactly the sort of outcome, reflecting a very narrow interpretation of biodiversity, that the working group was hoping to forestall.

We have here an unfortunately clear example of how focal species conservation can undermine efforts to achieve broader biodiversity conservation goals. Political leaders do not have the time or expertise to wade into the nuances of ecological trade-offs. It is much easier for them to latch onto a high profile species and use it as a proxy for biodiversity as a whole. In this case, the derailment of the entire land-use planning process (see Case Study 2) gave added impetus to the search for simple and quick land-use solutions.

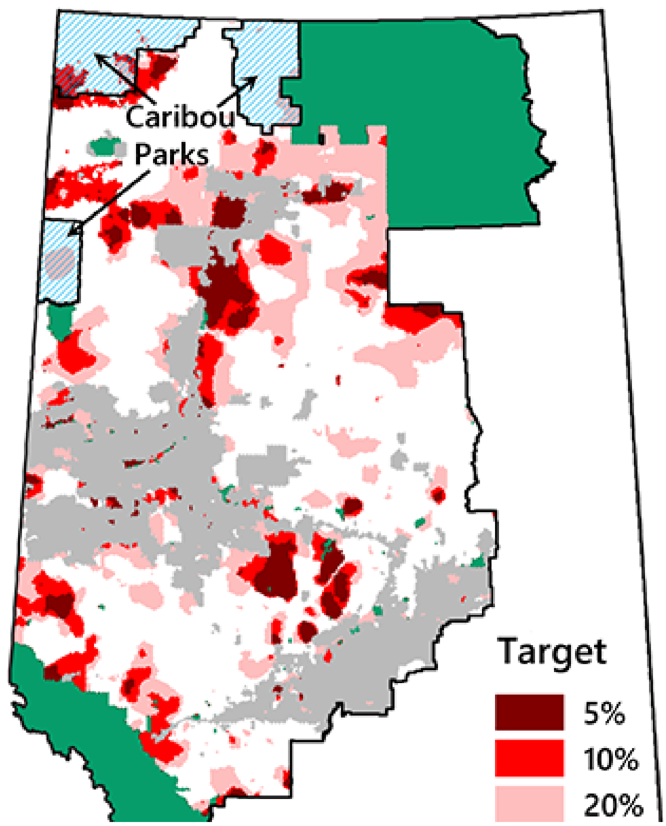


Fig. 11.31. A map overlay of the three caribou reserves proposed in the Alberta Caribou Range Plan (light blue) and the planning units selected in the base scenario with fine-filter elements added.

This is not to say that promoting the conservation of focal species is necessarily a bad strategy. Without a "poster child" to rally public attention and support, biodiversity conservation initiatives often fail to make progress. Much depends on local circumstances and there is no general rule to follow, except perhaps the old adage: be careful of what you ask for, you might get it.

CHAPTER XII

CONCLUSIONS

Conclusions

Conservation in Practice

In this final chapter, we will look back over what we have learned, highlighting a number of core themes about applied conservation. We will begin with a summary of the insights gained from the case studies. Then we will review the key factors associated with successful conservation initiatives. Finally, we will consider how effectiveness can be maximized at the personal level.

The case studies in the previous chapter do not necessarily reflect how conservation *should* be done, but how it typically *is* done. Much can be learned from these examples about how conservation theory is translated into conservation practice.

First, the case studies confirm that conservation is mainly about managing people rather than managing wild species per se. Cumulative disturbances are the main overarching concern. It is not the harvest of a single forest stand, or the landing of an individual fish, or the one-time application of a pesticide that creates a conservation problem. Concerns arise when these activities accumulate over time and space and when different types of activities occur in the same area.

Second, trade-off decisions are a central feature of conservation, and compromise solutions are the norm. This was a consistent theme in the case studies. In practice, we do not seek to maintain biodiversity, but to *limit the risk* to biodiversity from human activities to a socially acceptable level. This risk calculation varies from case to case and over time.

Third, aspirational conservation objectives derived from a sound ecological reference state are a critical component of effective conservation. They drive the search for innovative management solutions and guard against the acceptance of progressively diminished conservation outcomes over time. As demonstrated by the walleye example, pre-emptive compromises and shifting baselines are barriers to conservation.

Fourth, conservation decision making is not driven by value trade-offs alone. Science has a central role, providing the foundation for making informed, optimal choices. In the case studies, science was used to characterize threats, develop effective management approaches, predict outcomes under different management options, and monitor outcomes. Examples of decision making that are purely science or value driven do exist, but they are the exception rather than the rule.

Fifth, conservation decision making is hierarchical and involves different decision makers operating at different levels. In principle, legislation and high-level policies guide lower-level decision making concerned with implementation. However, the case studies illustrate that the decision hierarchy is not well integrated. Moreover, high-level policy tends to feature vague terms, such as "maintain" and "health," without defining what those terms mean (SARA being a notable exception). Consequently, the implementation of conservation measures entails interpreting higher-level policy in a case-specific context. This translation from policy to action is strongly influenced by local circumstances and by the interest, knowledge, and capacity of individuals operating at the base of the decision hierarchy. Thus, conservation is as much a bottom-up process as it is a top-down process.

Lastly, we see that conservation projects are not discrete entities. Individual projects are stages in an ever-evolving process subject to periods of relative stasis and periods of rapid change. Viewed over time, we can see that progress is being made (e.g., Al-Pac's new approach to forest harvesting). But new threats continue to emerge as well (e.g., the technological advances that made Al-Pac's mill economically viable in the first place). Like the mythical Sisyphus, who had to roll a large boulder up a hill each day only to have it roll down again, conservationists face a task that never ends.

Correlates of Success

The fundamental goal of conservation is to restore and then maintain species and ecosystems as they would be in their natural state. In practice, this goal is rarely achievable because compromises with other social objectives are usually required. Therefore, when assessing the effectiveness of conservation efforts, it is best to treat the natural state as an aspirational goal. Success is measured in terms of the progress made toward that ultimate goal.

The assessment of conservation success also needs to consider efficiency. Conservation resources are limited, so it is not only the amount of progress that is important, but the progress per unit of effort. A conservation initiative that achieves twice the conservation gain as another for the same level of input must be considered more successful. This is the basis of the optimal resource allocation concept.

Several fundamental themes concerning conservation success can be discerned:

1. **Effective leadership and collaboration.** Conservation initiatives generally require the collaborative contributions of many people. But having an individual, core group, or political champion who provides leadership and drive can make a big difference in the level of success achieved.

2. **Adequate resources.** Conservation initiatives require people with appropriate skills and funding for implementation. Many conservation issues languish because of inadequate resources.

3. **Authority for decision making and implementation.** Conservation success is highest when the threats being addressed fall within the purview of conservation practitioners. The recovery of walleye and the reintroduction of the swift fox are cases in point. In these cases, the conservation practitioners were operating within their domain of authority and could ensure that implementation would occur. The recovery of caribou provides a contrasting example. In this case, conservation practitioners had little influence over the decisions that really mattered (e.g., tenure allocation, forest harvest rates, infrastructure development). They knew what needed to be done but could not compel the necessary action.

When it comes to conservation at the ecosystem level, conservation practitioners only have meaningful management authority within protected areas. On the working landscape, management authority is divided among government departments with competing mandates, making it difficult to advance ecosystem-level conservation initiatives. As we saw in the regional planning case study, decision making at this scale is complex, politicized, and slow. Moreover, conservation objectives are easily diluted or treated superficially.

4. **An effective decision-making process.** Decision making is central to conservation. Conservation problems need to be prioritized so that efforts are focused where they will have the greatest benefit. We also need to determine which management actions out of the spectrum of possibilities are most effective. And we need to find optimal solutions to trade-offs among competing objectives. Thus, effective conservation depends on effective decision making, exemplified by the structured decision-making framework. Effective processes are in turn a function of sound institutional structures.

The application of structured decision-making principles was evident in the successful case studies. However, these real-world examples did not neatly follow the template outlined in Chapter 10. In each case, certain aspects of the process were emphasized over others because the planning challenges differed. For example, in the reserve design case study, considerable effort was needed in clarifying the objectives (i.e., determining what needed to be represented). In the swift fox reintroduction, the main challenge was assessing the effectiveness of the available management alternatives. In the Al-Pac case study, the planning team spent years developing innovative management approaches rather than choosing from among ineffective existing options. And in the walleye case study, the decisions turned on stakeholder-based preferences, informed by science.

The case studies also illustrated the effect of planning scope. As the scope of planning expanded, decision making became more complex and unwieldy, with more voices at the table, more objectives to balance, and more knowledge gaps to fill. The best outcomes were achieved in relatively simple and well-contained systems. The walleye case study provides a good example. Decision-making methods that were highly successful at the scale of individual lakes bogged down when later applied at the regional scale.

Of course, it is not always possible to break large problems into smaller manageable chunks. Many conservation problems can only be addressed through regional-scale decisions that address the root causes of conflict.

5. Public and political support. Conservation decisions generally reflect compromise solutions. The nature of this compromise is influenced by both the method of decision making and the level of support the competing values enjoy. As a general rule, the higher the level of conflict, the higher the level of support needed to make substantive conservation gains. The swift fox reintroduction and efforts to recover caribou provide contrasting examples (Fig. 12.1).

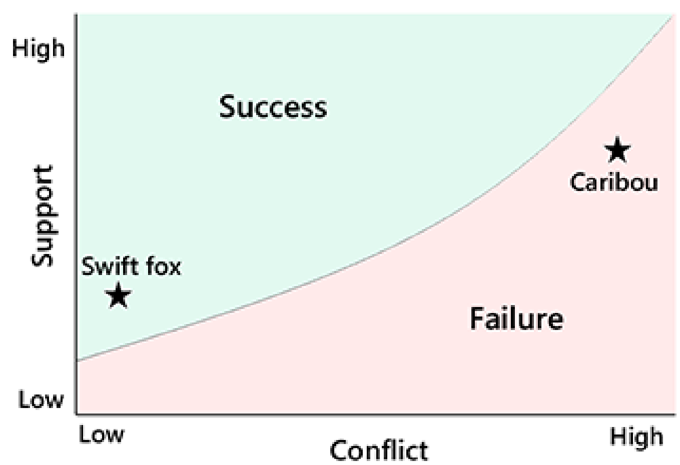


Fig. 12.1. Conservation outcomes are influenced by the level of conflict and by the level of public and political support. The swift fox reintroduction project had a low public profile, but also a low level of conflict. In contrast, caribou enjoyed a high level of public support, nevertheless conflicts with industry stymied recovery efforts.

Public support is critical, since it underpins the entire enterprise of conservation. Biodiversity is a public good and the vast majority of Canadian lands are publicly (Crown) owned. As discussed in Chapter 3, public support for conservation in Canada is strong and broad based. But it is not always effectively mobilized. Efforts to raise public awareness and engagement are often required to make progress.

It is also critical that public interests relating to conservation are properly represented in decision-making forums. Biodiversity has no voice of its own, so conservation concerns are sometimes overridden by the interests of local communities and industry. For example, in the caribou case study, we saw how local opposition caused the government to backtrack on caribou protection efforts in northwestern Alberta. It generally falls to conservation practitioners and other conservation proponents, both within and outside of government, to ensure that the public interest pertaining to biodiversity is given an effective voice. The application of conservation law, particularly SARA, can also be helpful in advancing the wider public interest.

The level of political support is another important factor in determining conservation outcomes. As we saw in the regional planning case study, the presence (or absence) of a political champion can make a big difference in how much progress is achieved. Conservation practitioners can generate political support for conservation by informing elected officials of conservation issues and providing workable solutions. But in many cases, the ruling party's ideology and policy platform will have an overriding influence on the positions it takes. Advancing conservation at this level requires political engagement and is mostly undertaken by environmental groups with a mandate for such activities.

Making a Difference

In this last section, we turn to effectiveness at the personal level. The practice of conservation requires a combination of broad and specialized knowledge—a so-called "T-shaped" skill set (Fig. 12.2; Schwartz et al. 2017). The stem of the T represents in-depth knowledge of a specific discipline. This knowledge is generally gained through formal study, typically in a subfield of biology but also in fields such as law, policy, social science, and economics.

The top bar of the T represents basic competence in a broad suite of ancillary fields relevant to working at the interface between science and policy (Jacobson and Duff 1998). Interdisciplinary knowledge and skills are typically acquired through self-directed study and work experience rather than university classes alone. This form of learning never really ends. The skills of greatest utility for advancing conservation include:

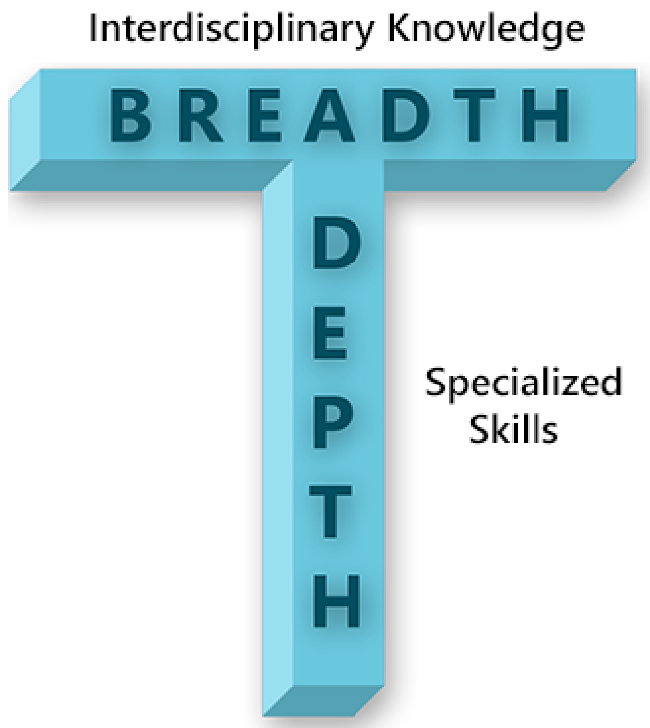


Fig. 12.2. Conservation practitioners should have a T-shaped skill set, where the stem of the T represents a core strength and the top bar represents basic competence in a suite of ancillary fields.

- **Communication ability.** Conservation invariably involves working with people, including colleagues, stakeholders, funders, elected officials, and the public. Therefore, the ability to communicate effectively, both orally and in written form, is a fundamental skill required by conservation practitioners. Effective communication also entails effective listening and a willingness to see the world from other points of view.
- **Decision-making skills.** Conservation practitioners should understand the principles of structured decision making and be able to apply them. Structured decision making should not be viewed as a rigid template, but as an effective way of thinking through problems that can be flexibly adapted to specific situations. Decision making also requires analytical skills for identifying problems, clarifying objectives, evaluating options, and so forth.
- **Management skills.** Besides making decisions, conservation practitioners need to oversee projects, facilitate groups, resolve conflicts, and accomplish various other tasks that fall under the general heading of management. Effective management requires good interpersonal and organizational skills as well as leadership ability.
- **Interdisciplinary knowledge.** To be effective, conservation practitioners need to understand the socio-political arena in which they are working. This includes institutional structures, relevant laws and policies, economic drivers, and the perspectives of the main stakeholders.

Effectiveness is also influenced by the way that personal resources, such as time, effort, and influence are allocated. Conservation practitioners should focus their efforts on initiatives that provide the greatest potential for advancing conservation, taking one's abilities and terms of employment into account. Finding an organization that provides a good fit with one's interests and abilities is part of this optimization process.

From our earlier discussion of the correlates of success, it may be tempting to conclude that it is best to focus on simple, well-contained conservation problems. But the calculus of effectiveness is deeper than that. The intractability of high-level initiatives is offset by their potential to benefit large numbers of species across wide spatial extents (Fig. 12.3). Even small conservation gains at this level can be important for maintaining overall biodiversity, and over the long term, these small gains can add up. Moreover, this is where the root causes of conservation problems are addressed. The upshot is that prioritization decisions are never easy, and they merit a thoughtful, structured process. Periodic reassessment is also important, because the parameters are constantly changing (e.g., conservation threats, conservation resources, knowledge gaps, and political windows of opportunity).

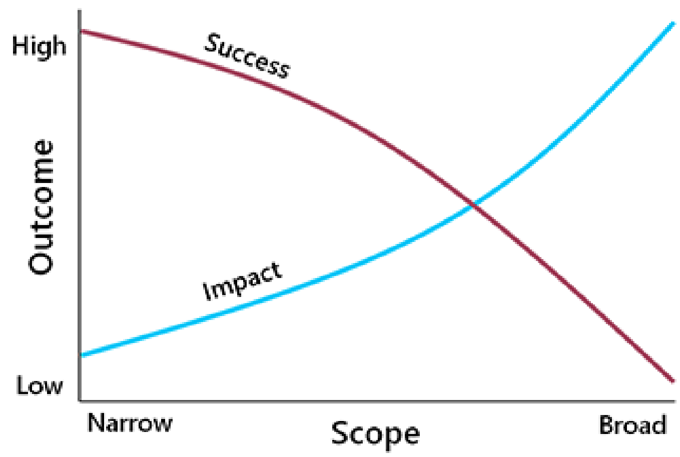


Fig. 12.3. Conservation initiatives with a broad scope tend to be complex and politicized, making it difficult to achieve desired conservation outcomes. However, such initiatives affect many species across large areas, so even small gains are important. Conservation initiatives with a narrow scope are more likely to achieve their objectives; however, their impact on overall biodiversity may be small.

Effectiveness should also be maximized within individual projects. Problems should be approached holistically and with an outcome-oriented mindset. The idea is to begin with the end in mind and let the final objective guide the choice of actions. This approach avoids the trap of "busy work," where much is being done but little is being accomplished.

With respect to specific practices, it is important to learn from the experience of others. Although the outcomes of conservation initiatives are not consistently reported, papers on conservation methods, tailored to specific species and problems, have slowly accumulated over the years. The collective body of knowledge is now quite large. A useful resource is the Conservation Evidence website, which provides a searchable database of conservation techniques based on over 5,000 published studies. This site provides a gateway to the literature.

In many cases, overcoming barriers to conservation requires innovative thinking and a willingness to be flexible. This can mean stepping out of one's comfort zone. The biologists hired by Al-Pac never expected to be government lobbyists, and the biologists managing walleye never expected to be workshop facilitators. Yet these activities were essential to success. The point is not that individual conservation practitioners can or should do everything, but that adaptability and a laser focus on outcomes are central to effectiveness.

Based on his long career in fisheries management, Carl Walters (2007) offers additional insights into what contributes to successful project outcomes, focusing on the role of individual conservation practitioners:

> The leaders have been people who (1) have a broad overview of the decision-making and implementation process, along with intimate knowledge of all the people and technical/administrative details involved in each step in the process; (2) are very well organized in terms of planning who, what, where, and when specific activities and actions are needed; (3) simply refuse to take no for an answer on the many occasions when contributors to the process offer excuses for inaction; and (4) are willing to devote their whole career, for extended periods of time (typically several years), to the implementation process. (p. 3)

Finally, it is important to recognize that conservation evolves slowly, in fits and starts. It is a long game that demands patience, persistence, and an optimistic “the glass is half full” outlook. Setbacks are many, and conservation practitioners need to be able to accept them, reassess, and move on. It is also important to recognize windows of opportunity when they arise and to act decisively when they do. In the final analysis, conservation practitioners cannot solve all of the world’s environmental problems, but we have an outsize effect relative to our numbers, and we can make a significant difference.

About the Author

Richard Schneider's career as a conservation biologist has evolved at the interface between conservation science and conservation policy. His professional training began at the University of Saskatchewan, where he obtained a BSc in biology, followed by a Doctor of Veterinary Medicine degree. In 1992, he obtained a PhD at the University of Guelph, studying wildlife epidemiology. This was followed by a postdoc at the University of Alberta that focused on the interaction between wolves, bison, and disease in northern Canada.

In the early stages of his career, Richard worked as a biological consultant, supporting government and industry conservation initiatives in different parts of Canada. The various species he worked on included martens in Newfoundland, Ontario, and Alaska; black-tailed prairie dogs in Saskatchewan; and burrowing owls, fishers, and caribou in Alberta.

In the early 2000s, Richard served as executive director for the northern Alberta chapter of the Canadian Parks and Wilderness Society. In this capacity, he became heavily involved in provincial land-use policy and planning, including the development of the Alberta Land-Use Framework. He also led a campaign for the establishment of new protected areas in northeastern Alberta (now the site of the world's largest boreal protected area network).

Since 2006, Richard has been a senior scientist with the Integrated Landscape Management Lab—now the Alberta Biodiversity Conservation Chair—at the University of Alberta. In this capacity, he has conducted applied research on industrial cumulative effects, the selection of protected areas in northern Alberta, and the recovery of woodland caribou. In recent years, his research has focused on the ecological effects of climate change and methods for climate adaptation. Richard has published numerous peer-reviewed papers as well as many lay publications and a book on ecosystem-based forest management (see Google Scholar).

Richard retired in 2020 and since then has been volunteering with Nature Alberta in various capacities.

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Glossary

A

Adaptation: (1) genetic changes driven by natural selection that increase individual fitness in a given environment; (2) physical or behavioural changes to accommodate changing conditions

Adaptive management: reducing decision-making uncertainties by monitoring and analyzing the outcomes of alternative management actions; learning by doing.

Alien species: a species living outside of its native range; also known as exotic species

Alleles: different forms of the same gene resulting from mutations of the DNA sequence

Allee effect: the positive relationship between the number of individuals in a population and the reproduction and survival of individuals

Alpha diversity: the species diversity that exists at a specific site (local-scale diversity)

Anthropocentric: an ethical perspective holding that humans are the most important elements of the world

Anthropogenic: originating from human activity

Assisted migration (or colonization): a climate adaptation measure that entails directly facilitating the movement of a species into suitable habitat outside of its historical range

Augmentation: the release of new individuals into an existing population to increase its size or genetic diversity

B

Beta diversity: a measure of the difference in species composition among sites

Bioamplification: the concentration of toxins in animals at the top of the food chain.

Biocentric: an ethical perspective holding that all life deserves equal moral consideration or has equal moral standing

Bioclimatic envelope model: a statistical model that describes the range of a species or the spatial distribution of an ecosystem as a function of the prevailing climatic conditions

Biodiversity: the variety of life in all its forms and at all levels of organization

Biological control: the release of a species, usually a predator or pathogen, to control a pest population

Biome: a major biological community that has formed in response to particular climatic conditions over a large geographic area

C

Carrying capacity: the number of individuals or biomass of a species that an ecosystem can support over the long term

Climate adaptation: adjusting to the effects of climate change

Climate mitigation: efforts to reduce the amount of future climate change, mostly through limits on the release of greenhouse gases

Climate refugia: areas where climatic conditions are expected to remain relatively stable or change very slowly despite progressive global warming

Climate velocity: the distance a climate envelope shifts per unit of time

Coarse-filter approach: (1) an approach to reserve design that entails protecting a representative sample of major ecosystem types; (2) a synonym for ecosystem-level conservation efforts

Community: an interacting group of species in a given location; the living members of an ecosystem

Competition: the interaction between individuals, groups, and species related to the acquisition of a limited resource

Connectivity: the degree to which a landscape facilitates or impedes the movement of animals and plants

Conservation easement: an approach to conservation in which a private landowner gives up certain legal rights concerning development, typically in exchange for tax benefits

Conservation practitioner: an individual with some form of conservation expertise working on applied conservation issues

Conservation triage: see optimal resource allocation

Contiguous: sharing a common border

Corridor: a special management zone intended to facilitate movement across barriers or to connect fragmented landscapes

COSEWIC: Committee on the Status of Endangered Wildlife in Canada; a body that oversees the assessment of species and makes recommendations regarding their listing as species at risk

Cost-benefit analysis: a component of decision making in which the positive and negative aspects of a proposed management action are tallied and compared, often in dollar terms.

Critical habitat: under the Species at Risk Act, the habitat necessary for the survival or recovery of a listed wildlife species

D

Demography: the study of population traits such as abundance, density, sex ratio, and rates of birth and death which together determine the dynamics of populations

Demographic stochasticity: random variations in the sex ratio and the rates of reproduction and death that can contribute to the decline of small populations

Density dependence: the regulation of population growth rates by factors related to population density, such as competition for food

Density independence: the regulation of population growth rates by factors unrelated to population density, such as catastrophic disturbances

Designatable units: a population, subspecies, or species that COSEWIC considers to be a discrete, evolutionarily significant unit for the purpose of assessment and listing

Dispersal: the movement of young plants and animals away from their parents

E

Easement: See conservation easement.

Ecological integrity: the degree to which an assemblage of organisms maintains its composition, structure, and function over time relative to a comparable assemblage that has been unaltered by human actions

Ecological niche: (1) the range of biotic and abiotic conditions necessary for species persistence; (2) the ecological role and position a species in an ecosystem

Ecological threshold: the point where an ecological indicator transitions from showing little response to increasing levels of a driver to a disproportionately large response

Ecosystem: a group of interacting organisms and the physical environment they inhabit

Ecosystem structure: the spatial arrangement of ecosystem components across multiple scales

Ecosystem composition: the variety and abundance of species in a given system

Ecosystem function: the ecological processes characteristic of living systems, such as succession, nutrient cycling, predation, and dispersal

Ecological reference state: the state of selected ecosystem indicators used as a target for management; often derived from the reconstructed preindustrial condition

Ecosystem management: a systems approach to resource management that places a priority on maintaining ecological structures and functions

Ecosystem services: benefits provided to human society from natural ecosystems

Ecotone: the transitional zone between two ecosystem types

Ecoregion: a coarse-scale unit of ecosystem classification featuring a relatively constant mix of environmental conditions and relatively distinct flora and fauna

Edge effects: the alterations in physical and biological conditions at the margins of habitat patches resulting from external influences

Endemic: (1) occurring in a place naturally; a native species; (2) describing a species as being found only in one geographic location, such as an island or nation

Environmental impact assessment: the part of a project approval process that examines the potential environmental consequences associated with a particular development

Environmental stochasticity: random variations in environmental conditions experienced by a population or community; a contributor to the decline of small populations

Eutrophication: an increase in the amount of nutrients in ecosystems as a result of human activities, especially agriculture

Evapotranspiration: the transfer of water to the atmosphere by evaporation from the soil and other surfaces and by transpiration from plants

Exotic species: see alien species

Extinction debt: species extinctions that are anticipated but delayed by biological lag effects

Extinction vortex: the synergistic action of demographic, genetic, and environmental processes that drive small populations to extinction

Extirpation: the extinction of a species from a specific geographic area; local extinction

F

Fitness: an individual's ability to grow, survive, and reproduce

Flagship species: a charismatic species that garners support for conservation

Focal species: a species of social significance that is the focus of management planning efforts

Frame: a mental construct that shapes the way we see the world and think about problems

G

Gamma diversity: the species diversity that exists across a broad region

Gene flow: the transfer of genetic material among populations arising from the movement of individuals

Gene pool: the total array of genes and alleles in a population

Genotype: the particular combination of alleles possessed by an individual

H

Habitat: the physical and biological environment using by an individual, population, or species

Habitat fragmentation: the transformation of contiguous habitat into a collection of small, isolated habitat patches through habitat loss

Heterogeneity: the state of being diverse in composition or character

Homogeneity: the state of being uniform in composition or character

Hot spots: areas identified as conservation priorities because of high species richness or high endemism

I

Inbreeding depression: the loss of fitness in small populations resulting from mating among related individuals

Indicator: a measurable system component or attribute used to assess the status of a system or a specific management objective

Industrial footprint: the cumulative modification of a natural landscape resulting from industrial activities.

Invasive species: species with high potential for expansion into new ecosystems; usually alien species

Irreplaceability: in reserve design, a description of planning units that cannot be substituted because they contain features not found elsewhere

IUCN: International Union for the Conservation of Nature

K

Keystone species: species that play a critical role in ecosystem function, above what would be expected from their abundance

L

Land ethic: a foundational concept underpinning Ecosystem-scale conservation first advanced by Aldo Leopold; nature is seen as an integrated system that needs to be managed as a whole

M

Maximum sustained yield: a harvest rate tuned to the natural demographics of a population such that production and harvest are maximal and non-declining

Metapopulation: a population that is divided into subpopulations as a result of natural or anthropogenic habitat fragmentation and which is linked by intermittent migration

Minimum viable population: the smallest population size able to persist over a defined interval at a given level of probability (e.g., a 99% chance of surviving over the next 100 years)

Mitigation measures: actions taken to prevent or reduce the adverse environmental effects of a project or other human activities

N

Natural: the state of a species or system unaffected by the activities of modern society

Natural disturbance model: the modification of industrial practices such that anthropogenic disturbances approximate the effects of natural disturbances

Natural range of variability (NRV): a statistical summary of the mean and variance of biotic elements and processes in a given area under natural conditions

Niche: see ecological niche

Normative: in science, research that is developed, presented, or interpreted based on an assumed preference for a particular policy

O

Offset: the restoration of an external site to compensate for unavoidable habitat losses from an industrial project

Old-growth forest: the stage in forest stand succession where natural senescence of the initial cohort of trees leads to individual tree replacement and increased structural complexity

Optimal resource allocation: in conservation, the efficient allocation of available resources to maximize conservation benefits

Opportunity cost: in the context of trade-off decisions, the values foregone when choosing one course of action over another

P

Path dependence: a situation where decisions made in the past constrain what is feasible in the present

Perturbation: A deviation of a system or process from its normal state or path, caused by an outside influence

Phenotype: the morphological, physiological, and biochemical characteristics of an individual arising from the expression of its genotype in a particular environment

Population: a group of individuals that live in a particular geographical area and normally breed with one another

Population viability analysis: a modeling approach for predicting a population's likelihood of persistence on the basis of demographic parameters

Precautionary principle: a management concept which states that measures to prevent environmental degradation should not be delayed because of a lack of full scientific certainty

Preindustrial baseline: the environmental conditions prior to industrial development, commonly used as a proxy for the natural state in conservation initiatives

R

Ratchet effect: see shifting baseline

Reclamation: the repair of damage to a degraded site without necessarily recreating the original ecosystem

Reintroduction: The release of captive bred or Wild-collected individuals into a part of their historical range where they no longer occur

Remote sensing: the use of high-resolution imagery from satellites and aircraft to study the earth's surface, often for the purpose of landscape classification and monitoring

Replacement cost: the estimated value of an ecosystem service based on the cost of replacing it with a human-based alternative

Rescue effect: the process whereby extirpation of small populations is prevented through immigration from other subpopulations

Reserve: a synonym for protected area

Resilience: the ability of a species or ecosystem to return to its original state after a disturbance

Resource selection function: a statistical model used to explain and predict habitat selection by individuals at the local scale

Restoration: the process of returning a degraded ecosystem to its natural state

Richness: the number of different species at a site or within a region

S

Salvage logging: the removal of standing dead trees after fire or other disturbance

Species at Risk Act (SARA): federal legislation governing the recovery of Canadian species at risk

Scenarios: in planning applications, alternative visions of how the future might unfold, chosen to foster learning about the management of a system

Sensitivity analysis: a modelling technique used to identify the most influential variables in a system as well as the main sources of uncertainty

Shifting mosaic: the complex, everchanging landscape pattern created by the turnover of habitat patches from disturbance and succession

Shifting baseline: where conservation objectives are reset each generation based on existing conditions, locking in losses that have already occurred

Silviculture: the practice of controlling the establishment, growth, composition, and quality of forests to meet specified management objectives

Sink population: a population existing in suboptimal conditions that must rely on immigration to remain viable

SLOSS: a debate concerning the design of protected areas focused on the relative merits of having a single large or several small reserves

Social licence: the standards that organizations must meet to achieve acceptance of their operations by local communities, stakeholders, and the public

Source population: a population that is intrinsically viable and can serve as a source of emigrants to other areas

Species: in conservation applications, a group of individuals capable of interbreeding under natural conditions

Species at risk: species listed as threatened or endangered under federal or provincial law and subject to special protection and recovery planning

Stochasticity: random variation in a ecological or environmental processes, such as weather, natural disturbances, and reproduction

Succession: the gradual and sequential process of change in ecosystem composition and structure following natural or anthropogenic disturbance

Sustainable development: development that meets the needs of the present without compromising the ability of future generations to meet their own needs

Sustainable forest management: forestry designed to maintain and enhance the long-term health of forest ecosystems while providing benefits to present and future generations

Systematic conservation planning: a framework for reserve design featuring optimization methods for achieving comprehensive representation of biodiversity elements and the maximization of other design features

T

Telemetry: the collection of information using a remote transmitter, such as an animal collar fitted with a GPS unit

Trade-off: the necessity of giving up one thing to get something else when a constraint or incompatibility prevents the simultaneous achievement of multiple objectives

Tragedy of the Commons: a resource management problem in which the users of a shared resource end up depleting it through the narrow pursuit of self interest

Translational ecology: an approach in which conservation practitioners, stakeholders, and decision makers work together to develop research that addresses the practical dimensions of an environmental problem

Translocation: the intentional release of organisms from one area into another

Triad approach: a form of landuse zonation that includes a sustainable use zone, an intensive management zone, and a fully protected zone

U

Umbrella species: species with large home ranges and broad habitat requirements that can serve as surrogates for other species in habitat management initiatives

Utilitarian perspective: a viewpoint that emphasizes the direct benefits of biological systems and components to humans

V

Values: deeply held beliefs about what is desirable, right, and appropriate

Bibliography

AAFC. 2003. Greencover Canada's Land Conversion Component. Agriculture and Agri-Food Canada, Ottawa, ON.

AAFC. 2018. An Overview of the Canadian Agriculture and Agri-Food System 2017. Agriculture and Agri-Food Canada, Ottawa, ON.

Aakala, T., C. Remy, D. Arseneault, H. Morin, and M. Girardin. 2023. Millennial-scale disturbance history of the boreal zone. Pages 53-87 in Boreal Forests in the Face of Climate Change, edited by M. Girona, S. Gauthier, H. Morin, and Y. Bergeron. Advances in Global Change Research, Volume 74.

Aarts, G., J. Fieberg, and J. Matthiopoulos. 2012. Comparative interpretation of count, presence–absence and point methods for species distribution models. Methods in Ecology and Evolution 3:177-187.

ABMI. 2017a. Alberta Biodiversity Monitoring Institute Annual Report 16/17. Alberta Biodiversity Monitoring Institute, Edmonton, AB.

ABMI. 2017b. Terrestrial Field Data Collection Protocols (Abridged Version). Alberta Biodiversity Monitoring Institute, Edmonton, AB.

ABMI. 2020. Status of Land Cover and Biodiversity in the Al-Pac Forest Management Agreement Area: Five-Year Update. Alberta Biodiversity Monitoring Institute, Edmonton, AB.

Abrahms, B., D. DiPietro, A. Graffis, and A. Hollander. 2017. Managing biodiversity under climate change: challenges, frameworks, and tools for adaptation. Biodiversity and Conservation 26:2277-2293.

Adde, A., C. Casabona, M. Mazerolle, M. Darveau and S. Cumming. 2021. Integrated modeling of waterfowl distribution in western Canada using aerial survey and citizen science (eBird) data Ecosphere 12:e03790.

Addison, P., L. Rumpff, S. Bau, J. Carey, and Y. Chee. 2013. Practical solutions for making models indispensable in conservation decision-making. Diversity and Distributions 19:490-502.

AEP. 1998. The Boreal Forest Natural Region of Alberta – Report to the Special Places 2000 Provincial Coordinating Committee. Alberta Environmental Protection, Edmonton, AB.

AEP 2020. Citizen science principles of good practice. Alberta Environment and Parks, Edmonton, AB.

AER. 2020. Inactive Well Compliance Program: Year Three and Four Final Report. Alberta Energy Regulator, Edmonton, AB.

AFGA. 2018. Redefining Fisheries Management in Alberta. Alberta Fish and Game Association, online report: www.afga.org.

AFPA. 2006. Alberta Forest Usage Survey Results. Alberta Forest Products Association, Edmonton, AB.

AGC. 2013. Report of the Commissioner of the Environment and Sustainable Development. Chapter 2. Auditor General of Canada, Ottawa, ON.

Aklin, M. and J. Urpelainen. 2014. Perceptions of scientific dissent undermine public support for environmental policy. Environmental Science & Policy 38:173-177.

Alava, J., P. Ross, and F. Gobas. 2016. Food web bioaccumulation model for resident killer whales from the Northeastern Pacific Ocean as a tool for the derivation of PBDE-sediment quality guidelines. Archives of Environmental Contamination and Toxicology 70:155-168.

Allan, R. 1997. Introduction: mining and metals in the environment. Journal of Geochemical Exploration 58:95-100.

Alldredge, J. and J. Griswold. 2006. Design and analysis of resource selection studies for categorical resource variables. Journal of Wildlife Management 70:337-346.

Allen, E. and L. Humble. 2002. Nonindigenous species introductions: a threat to Canada's forests and forest economy. Canadian Journal of Plant Pathology 24:103-110.

Al-Pac. 2007. Alberta-Pacific Forest Management Plan (Revised) September 2007. Alberta-Pacific Forest Industries, Boyle, AB.

Al-Pac. 2015. Alberta-Pacific FMA Area: 2015 Forest Management Plan. Alberta-Pacific Forest Industries, Boyle, AB.

ALUS. 2023. What We Do. Downloaded Feb. 20, 2023 from the Alternative Land Use Services Canada website: alus.ca/what-we-do/

Anderson, J., G. Armstrong, M. Luckert, and W. Adamowicz. 2012. Optimal zoning of forested land considering the contribution of exotic plantations. Mathematical and Computational Forestry & Natural Resource Sciences 4:92-104.

Anderson, M. and C. Ferree. 2010. Conserving the stage: climate change and the geophysical underpinnings of species diversity. PLoS One 5:e11554.

Anderson, R. 2006. Evolution and origin of the Central Grassland of North America: climate, fire, and mammalian grazers. The Journal of the Torrey Botanical Society 133:626-647.

Andison, D. and K. McCleary. 2014. Detecting regional differences in within-wildfire burn patterns in western boreal Canada. The Forestry Chronicle 90:59-69.

Ando, A. and M. Mallory. 2012. Optimal portfolio design to reduce climate-related conservation uncertainty in the Prairie Pothole Region. Proceedings of the National Academy of Sciences 109:6484-6489.

Andrew, M., M. Wulder, and J. Cardille. 2014. Protected areas in boreal Canada: a baseline and considerations for the continued development of a representative and effective reserve network. Environmental Reviews 22:135-160.

Anielski, M. and S. Wilson. 2002. Counting Canada's Natural Capital: Assessing the Real Value of Canada's Boreal Ecosystems. Pembina Institute, Calgary, AB.

AP. 2014. Natural Regions & Subregions of Alberta: A Framework for Alberta Parks. Alberta Parks, Edmonton, AB.

Araujo, M. and A. Guisan. 2006. Five (or so) challenges for species distribution modelling. Journal of Biogeography 33:1677-1688.

Araujo, M. and A. Peterson. 2012. Uses and misuses of bioclimatic envelope modeling. Ecology 93:1527-1539.

Archer, D. and V. Brovkin. 2008. The millennial atmospheric lifetime of anthropogenic CO_2. Climatic Change 90:283-297.

Arlettaz, R., M. Schaub, J. Fournier, T. Reichlin, and A. Sierro. 2010. From publications to public actions: when conservation biologists bridge the gap between research and implementation. BioScience 60:835-842.

Armitage, D., F. Berkes, A. Dale, E. Kocho-Schellenberg, and E. Patton. 2011. Co-management and the co-production of knowledge: learning to adapt in Canada's Arctic. Global Environmental Change 21:995-1004.

Armstrong, D. and P. Seddon. 2008. Directions in reintroduction biology. Trends in Ecology & Evolution 23:20-25.

Armstrong, G., S. Cumming, and W. Adamowicz. 1999. Timber supply implications of natural disturbance management. The Forestry Chronicle 75:497-504.

Arponen, A. 2012. Prioritizing species for conservation planning. Biodiversity and Conservation 21:875-893.

Ashcroft, M., J. Gollan, D. Warton, and D. Ramp. 2012. A novel approach to quantify and locate potential microrefugia using topoclimate, climate stability, and isolation from the matrix. Global Change Biology 18:1866-1879.

Auerbach, N., K. Wilson, A. Tulloch, J. Rhodes, and J. Hanson. 2015. Effects of threat management interactions on conservation priorities. Conservation Biology 29:1626-1635.

Backstrom, A., G. Garrard, R. Hobbs, and S. Bekessy. 2018. Grappling with the social dimensions of novel ecosystems. Frontiers in Ecology and the Environment 16:109-117.

Bailey, A., D. McCartney, and M. Schellenberg. 2010. Management of Canadian Prairie Rangeland. Agriculture and Agri-Food Canada, Ottawa, ON.

Baker, W. 1994. Restoration of landscape structure altered by fire suppression. Conservation Biology 8:763-769.

Ball, M., G. Somers, J. Wilson, R. Tanna, and C. Chung. 2013. Scale, assessment components, and reference conditions: issues for cumulative effects assessment in Canadian watersheds. Integrated Environmental Assessment and Management 9:370-379.

Balshi, M., A. McGuire, P. Duffy, M. Flannigan, and J. Walsh. 2009. Assessing the response of area burned to changing climate in western boreal North America using a Multivariate Adaptive Regression Splines (MARS) approach. Global Change Biology 15:578-600.

Bankes, N., S. Mascher, and M. Olszynski. 2014. Can environmental laws fulfill their promise? Stories from Canada. Sustainability 6:6024-6048.

Barber, Q., S. Nielsen, and A. Hamann. 2016. Assessing the vulnerability of rare plants using climate change velocity, habitat connectivity, and dispersal ability: a case study in Alberta, Canada. Regional Environmental Change 16:1433-1441.

Barbour, M., N. Poff, R. Norris, and J. Allan. 2008. Perspective: communicating our science to influence public policy. Perspective 27:562-569.

Barnas, K., S. Katz, D. Hamm, M. Diaz, C. Jordan. 2015. Is habitat restoration targeting relevant ecological needs for endangered species? Using Pacific Salmon as a case study. Ecosphere 6:110.

Bartzen, B., K. Dufour, R. Clark, and F. Caswell. 2010. Trends in agricultural impact and recovery of wetlands in prairie Canada. Ecological Applications 20:525-538.

Battin, J., M. Wiley, M. Ruckelshaus, R. Palmer, and E. Korb. 2007. Projected impacts of climate change on salmon habitat restoration. Proceedings of the National Academy of Sciences 104:6720-6725.

Baum, J. and S. Fuller. 2016. Canada's Marine Fisheries: Status, Recovery Potential and Pathways to Success. Oceana, online report: www.oceana.ca.

Baumgartner, F., B. Jones, and P. Mortensen. 2014. Punctuated equilibrium theory: explaining stability and change in public policymaking. Pages 59-104 in Theories of the Policy Process, edited by P. Sabatier and C. Weible. Westview Press, Boulder, CO.

Bayne, E., S. Boutin, and R. Moses. 2008. Ecological factors influencing the spatial pattern of Canada lynx relative to its southern range edge in Alberta, Canada. Canadian Journal of Zoology 86:1189-1197.

BCARDC. 2021. Reference Guide: The Canada-British Columbia Environmental Farm Plan Program. BC Agricultural Research & Development Corporation, Abbotsford, BC.

BCEAO. 2012. Environmental Assessment Office Statistics. BC Environmental Assessment Office, online data: www.eao.gov.bc.ca/statistics.html.

BCMOE. 2009. Conservation Framework: Conservation Priorities for Species and Ecosystems. B.C. Ministry of the Environment. Victoria, B.C.

BCMOF. 2012. Ecosystem Based Management on B.C.'s Central and North Coast (Great Bear Rainforest): Implementation Update Report. BC Ministry of Forests, Victoria, BC.

Bedford, F., R. Whittaker, and J. Kerr. 2012. Systemic range shift lags among a pollinator species assemblage following rapid climate change. Botany 90:587-597.

Beier, P. 2012. Conceptualizing and designing corridors for climate change. Ecological Restoration 30:312-319.

Beier, P. and B. Brost. 2010. Use of land facets to plan for climate change: conserving the arenas, not the actors. Conservation Biology 24:701-710.

Beier, P., W. Spencer, R. Baldwin, and B. McRae. 2011. Toward best practices for developing regional connectivity maps. Conservation Biology 25:879-892.

Belisle, A., S. Gauthier, D. Cyr, Y. Bergeron, and H. Morin. 2011. Fire regime and old-growth boreal forests in central Quebec, Canada: an ecosystem management perspective. Silva Fennica 45:889-908.

Benkman, C. 1993. Decline of the red crossbill of Newfoundland. American Birds 47:225-229.

Bergeron, Y., D. Cyr, M. Girardin, and C. Carcaillet. 2011. Will climate change drive 21st century burn rates in Canadian boreal forest outside of its natural variability: collating global climate model experiments with sedimentary charcoal data. Inter- national Journal of Wildland Fire 19:1127-1139.

Bergeron, Y., P. Drapeau, S. Gauthier, and N. Lecomte. 2007. Using knowledge of natural disturbances to support sustainable forest management in the northern Clay Belt. The Forestry Chronicle 83:326-337.

Bergeron, Y. and N. Fenton. 2012. Boreal forests of eastern Canada revisited: old growth, nonfire disturbances, forest succession, and biodiversity. Botany 90:509-523.

Berkes, F. 2012. A story of caribou and social learning. Pages 125-143 in Sacred Ecology: Traditional Ecological Knowledge and Resource Management. Routledge. New York, NY.

Berkes, F., J. Colding, and C. Folke. 2000. Rediscovery of traditional ecological knowledge as adaptive management. Ecological Applications 10:1251-1262.

Berteaux, D., S. de Blois, J. Angers, J. Bonin, and N. Casajus. 2010. The CC-Bio Project: studying the effects of climate change on Quebec biodiversity. Diversity 2:1181-1204.

Biggs, D., N. Abel, A. Knight, A. Leitch, and A. Langston. 2011. The implementation crisis in conservation planning: could “mental models” help? Conservation Letters 4:169-183.

Bird, S. and K. Hodges. 2017. Critical habitat designation for Canadian listed species: slow, biased, and incomplete. Environmental Science & Policy 71:1-8.

Bjorkman, A. and M. Vellend. 2010. Defining historical baselines for conservation: ecological changes since European settlement on Vancouver Island, Canada. Conservation Biology 24:1559-1568.

Boakes, E., N. Isaac, R. Fuller, G. Mace, and P. McGowan. 2018. Examining the relationship between local extinction risk and position in range. Conservation Biology 32:229-239.

Bottrill, M., L. Joseph, J. Carwardine, M. Bode, and C. Cook. 2008. Is conservation triage just smart decision making? Trends in Ecology & Evolution 23:649-654.

Bouchard, M. and D. Pothier. 2010. Spatiotemporal variability in tree and stand mortality caused by spruce budworm outbreaks in eastern Quebec. Canadian Journal of Forest Research 40:86-94.

Bouchard, M., D. Pothier, and S. Gauthier. 2008. Fire return intervals and tree species succession in the North Shore region of eastern Quebec. Canadian Journal of Forest Research 38:1621-1633.

Boucher, Y., D. Arseneault, L. Sirois, and L. Blais. 2009. Logging pattern and landscape changes over the last century at the boreal and deciduous forest transition in Eastern Canada. Landscape Ecology 24:171-184.

Boukal, D. and L. Berec. 2002. Single-species models of the Allee effect: extinction boundaries, sex ratios and mate encounters. Journal of Theoretical Biology 218:375-394.

Boulanger, J., A. Gunn, J. Adamczewski, and B. Croft. 2011. A data-driven demographic model to explore the decline of the Bathurst caribou herd. The Journal of Wildlife Management 75:883-896.

Boutilier, R. 2014. Frequently asked questions about the social licence to operate. Impact Assessment and Project Appraisal 32:263-272.

Bowlby, H. and A. Gibson. 2011. Reduction in fitness limits the useful duration of supplementary rearing in an endangered salmon population. Ecological Applications 21:3032-3048.

Boyce, M. 1992. Population viability analysis. Annual Review of Ecology and Systematics 23:481-497.

Boyce, M., C. Johnson, E. Merrill, S. Nielsen, and E. Solberg. 2016. Can habitat selection predict abundance? Journal of Animal Ecology 85:11-20.

Boyce, M., A. Sinclair, and G. White. 1999. Seasonal compensation of predation and harvesting. Oikos 87:419-426.

Boyd, D.R. 2003. Unnatural Law: Rethinking Canadian Environmental Law and Policy. UBC Press, Vancouver, BC.

Brito, D. and C. de Viveiros Grelle. 2004. Effectiveness of a reserve network for the conservation of the endemic marsupial Micoureus travassosi in Atlantic Forest remnants in southeastern Brazil. Biodiversity & Conservation 13:2519-2536.

Brownsey, K. and J. Rayner. 2009. Integrated land management in Alberta: from economic to environmental integration. Policy and Society 28:125-137.

Brulle, R. and C. Jenkins 2010. Civil society and the environment: understanding the dynamics and impacts of the U.S. environmental movement. Pages 73-102 in Good Cop/Bad Cop: Environmental NGOs and Their Strategies Toward Business, edited by T. Lyon. RFF Press, Washington, DC.

Brunner, R. and T. Clark. 1997. A practice-based approach to ecosystem management. Conservation Biology 11:48-58.

Brussard, P. and J. Tull. 2007. Conservation biology and four types of advocacy. Conservation Biology 21:21-24.

Bucher, E.H. 1992. The causes of extinction of the passenger pigeon. Current Ornithology 9:1-36.

Buckley, R. 2016. Triage approaches send adverse political signals for conservation. Frontiers in Ecology and Evolution 4:39.

Bujold, R., M. Simon, D. Anderson, D. Dobell, and T. Hayes. 2018. Final Report of the National Advisory Panel on Marine Protected Area Standards. Fisheries and Oceans Canada, Ottawa, ON.

Burnett, A. 2003. A Passion for Wildlife: The History of the Canadian Wildlife Service. UBC Press, Vancouver, BC.

Burton, A., E. Neilson, D. Moreira, A. Ladle, and R. Steenweg. 2015. Wildlife camera trapping: a review and recommendations for linking surveys to ecological processes. Journal of Applied Ecology 52:675-685.

Burton, P., C. Messier, W. Adamowicz, and T. Kuuluvainen. 2006. Sustainable management of Canada's boreal forests: progress and prospects. Ecoscience 13:234-248.

BWG. 2015. 2020 Biodiversity Goals and Targets for Canada. Biodiversity Working Group, Environment Canada, Ottawa, ON.

Cabin, R. 2007. Science-driven restoration: a square grid on a round earth? Restoration Ecology 15:1-7.

Callaghan, C., J. Martin, R. Major, and R. Kingsford. 2018. Avian monitoring – comparing structured and unstructured citizen science. Wildlife Research 45:176-184.

Callaghan, C., A. Poore, M. Hofmann, C. Roberts, and H. Pereira. 2021. Scientific Reports 11:19073.

Camaclang, A., M. Maron, T. Martin, and H. Possingham. 2015. Current practices in the identification of critical habitat for threatened species. Conservation Biology 29:482-492.

Cameron, E. and E. Bayne. 2009. Road age and its importance in earthworm invasion of northern boreal forests. Journal of Applied Ecology 46:28-36.

Campbell, C., I. Campbell, C. Blyth, and J. McAndrews. 1994. Bison extirpation may have caused aspen expansion in western Canada. Ecography 17:360-362.

CAPP. 2017. Statistical Handbook for Canada's Upstream Petroleum Industry. Canadian Association of Petroleum Producers, Calgary, AB.

Carbyn, L. 1998. Update COSEWIC Status Report on the Swift Fox (Vulpes velox) in Canada. Canadian Wildlife Service, Western Region, Edmonton, AB.

Carlson, M. and W. Kurz. 2007. Approximating natural landscape pattern using aggregated harvest. Canadian Journal of Forest Research 37:1846-1853.

Carmichael, J., R. Brulle, and J. Huxster. 2017. The great divide: understanding the role of media and other drivers of the partisan divide in public concern over climate change in the USA, 2001–2014. Climatic Change 141:599-612.

Carroll, C., B. Hartl, G. Goldman, D. Rohlf, and A. Treves. 2017. Defending the scientific integrity of conservation-policy processes. Conservation Biology 31:967-975.

Carroll, C., R. Noss, P. Paquet, and N. Schumaker. 2003. Use of population viability analysis and reserve selection algorithms in regional conservation plans. Ecological Applications 13:1773-1789.

Carr, A., P. Weedon, and E. Cloutis. 2004. Climate Change Implications in Saskatchewan's Boreal Forest Fringe and Surrounding Agricultural Areas. Saskatchewan Forest Centre, Prince Albert, SK.

Carwardine, J., T. Martin, J. Firn, R. Reyes, and S. Nicol. 2019. Priority Threat Management for biodiversity conservation: a handbook. Journal of Applied Ecology 56:481-490.

Carwardine, J., K. Wilson, M. Watts, A. Etter, and C. Klein. 2008. Avoiding costly conservation mistakes: the importance of defining actions and costs in spatial priority setting. PLoS One 3:e2586.

Castrilli, J. 2010. Wanted: a legal regime to clean up orphaned /abandoned mines in Canada. McGill International Journal of Sustainable Development Law and Policy 6:109-141.

Caughley, G. 1994. Directions in conservation biology. Journal of Animal Ecology 63:215-244.

CCEA. 2017. Conservation Areas Reporting and Tracking System. Canadian Council on Ecological Areas, online data: www.ccea.org.

CCFM. 1992. Canada Forest Accord. Canadian Council of Forest Ministers, Ottawa, ON.

CCFM. 2003. Defining Sustainable Forest Management in Canada: Criteria and Indicators 2003. Canadian Council of Forest Ministers, Ottawa, ON.

CCME. 2009. Regional Strategic Environmental Assessment in Canada: Principles and Guidance. Canadian Council of Ministers of the Environment, Winnipeg, MB.

CCME. 2014. Canadian Council of Ministers of the Environment. Canadian Council of Ministers of the Environment, Winnipeg, MB.

CCRM. 2010. Canadian Biodiversity: Ecosystem Status and Trends 2010. Canadian Council of Resource Ministers, Ottawa, ON.

CCRM. 2014. 2012 Canadian Nature Survey: Awareness, Participation, and Expenditures in Nature-based Recreation, Conservation, and Subsistence Activities. Canadian Council of Resource Ministers, Ottawa, ON.

Ceballos, G., P. Ehrlich, and R. Dirzo. 2017. Biological annihilation via the ongoing sixth mass extinction signaled by vertebrate population losses and declines. Proceedings of the National Academy of Sciences 114:E6089-E6096.

CEMA. 2008. Terrestrial Ecosystem Management Framework for the Regional Municipality of Wood Buffalo. Cumulative Environmental Management Association, Fort McMurray, AB.

CESCC. 2022. Wild Species 2020: The General Status of Species in Canada. Canadian Endangered Species Conservation Council, Ottawa, ON.

CFIA. 2008. Invasive Alien Plants in Canada: Technical Report. Canadian Food Inspection Agency, Ottawa, ON.

CFS. 2022. The State of Canada's Forests: Annual Report 2022. Canadian Forest Service, Ottawa, ON.

Chan, K., T. Satterfield, and J. Goldstein. 2012. Rethinking ecosystem services to better address and navigate cultural values. Ecological Economics 74:8-18.

Channell, R. and M. Lomolino. 2000. Trajectories to extinction: spatial dynamics of the contraction of geographical ranges. Journal of Biogeography 27:169-179.

Chapleau, F., C. Findlay, and E. Szenasy. 1997. Impact of piscivorous fish introductions on fish species richness of small lakes in Gatineau Park, Quebec. Ecoscience 4:259-268.

Charlesworth, D. and J. Willis. 2009. The genetics of inbreeding depression. Nature Reviews Genetics 10:783-796.

Christensen, N., A. Bartuska, J. Brown, S. Carpenter, and C. D'Antonio. 1996. The report of the Ecological Society of America committee on the scientific basis for ecosystem management. Ecological Applications 6:665-691.

Clark, P., J. Shakun, S. Marcott, A. Mix, and M. Eby. 2016. Consequences of twenty-first-century policy for multi-millennial climate and sea-level change. Nature Climate Change 6:360-369.

Clark, T. 2001. Developing policy-oriented curricula for conservation biology: professional and leadership education in the public interest. Conservation Biology 15:31-39.

Clark, T. 2002. The Policy Process: A Practical Guide for Natural Resource Professionals. Yale University Press, New Haven, CT.

Clevenger, A. and M. Huijser. 2011. Wildlife Crossing Structure Handbook: Design and Evaluation in North America. US Department of Transportation, Federal Highway Administration, Lakewood, CO.

CMP. 2013. Open Standards for the Practice of Conservation. Conservation Measures Partnership, online report: http://cmp-openstandards.org.

Cochran-Biederman, J., K. Wyman, W. French, and G. Loppnow. 2015. Identifying correlates of success and failure of native freshwater fish reintroductions. Conservation Biology 29:175-186.

Coissac, E., P. Hollingsworth, S. Lavergne, and P. Taberlet. 2016. From barcodes to genomes: extending the concept of DNA barcoding. Molecular Ecology 25:1423-1428.

Colla, S. 2016. Status, threats and conservation recommendations for wild bumble bees (Bombus spp.) in Ontario, Canada: a review for policymakers and practitioners. Natural Areas Journal 36:412-426.

Conklin, B. and L. Graham. 1995. The shifting middle ground: Amazonian Indians and eco-politics. American Anthropologist 97:695-710.

Connelly, R. 2011. Canadian and international EIA frameworks as they apply to cumulative effects. Environmental Impact Assessment Review 31:453-456.

Conroy, M. and J. Peterson. 2013. Decision Making in Natural Resource Management. Wiley-Blackwell, West Sussex, UK.

Converse, S., C. Moore, M. Folk, and M. Runge. 2013. A matter of tradeoffs: reintroduction as a multiple objective decision. The Journal of Wildlife Management 77:1145-1156.

Cook, C., K. de Bie, D. Keith, and P. Addison. 2016. Decision triggers are a critical part of evidence-based conservation. Biological Conservation 195:46-51.

Cook, C., M. Mascia, M. Schwartz, H. Possingham, and R. Fuller. 2013. Achieving conservation science that bridges the knowledge—action boundary. Conservation Biology 27:669-678.

Cook, J., N. Oreskes, P. Doran, W. Anderegg, and B. Verheggen. 2016. Consensus on consensus: a synthesis of consensus estimates on human-caused global warming. Environmental Research Letters 11:048002.

Coops, N., J. Timko, M. Wulder, J. White, and S. Ortlepp. 2008. Investigating the effectiveness of mountain pine beetle mitigation strategies. International Journal of Pest Management 54:151-165.

Corlett, R. and D. Westcott. 2013. Will plant movements keep up with climate change? Trends in Ecology & Evolution 28:482-488.

Cortus, B., S. Jeffrey, J. Unterschultz, and P. Boxall. 2011. The economics of wetland drainage and retention in Saskatchewan. Canadian Journal of Agricultural Economics 59:109-126.

COSEWIC. 2006. COSEWIC Assessment and Update Status Report on the Burrowing Owl, Athene cunicularia, in Canada. Committee on the Status of Endangered Wildlife in Canada. Ottawa, ON.

COSEWIC. 2007. COSEWIC Assessment and Update Status Report on the Peregrine Falcon (Falco pereginus) in Canada. Committee on the Status of Endangered Wildlife in Canada, Ottawa, ON.

COSEWIC. 2016. COSEWIC Assessment and Status Report on the Red Crossbill Percna Subspecies Loxia Curvirostra Percna in Canada. Committee on the Status of Endangered Wildlife in Canada. Ottawa, ON.

Costello, M., R. May, and N. Stork. 2013. Can we name Earth's species before they go extinct? Science 339:413-416.

Cote, P., R. Tittler, C. Messier, D. Kneeshaw, and A. Fall. 2010. Comparing different forest zoning options for landscape-scale management of the boreal forest: possible benefits of the TRIAD. Forest Ecology and Management 259:418-427.

Cowan, W., W. Mackasey, and J. Robertson. 2010. The Policy Framework in Canada for Mine Closure and Management of Long-term Liabilities: A Guidance Document. National Orphaned/Abandoned Mines Initiative, Ottawa, ON.

Craig, R. 2010. Stationarity is dead—long live transformation: five principles for climate change adaptation law. Harvard Environmental Law Review 34:9-75.

Cranstone, D. 2002. A History of Mining and Mineral Exploration in Canada and Outlook For The Future. Natural Resources Canada, Ottawa, ON.

Crowe, K. and W. Parker. 2008. Using portfolio theory to guide reforestation and restoration under climate change scenarios. Climatic Change 89:355-370.

Cullingham, C., J. Cooke, S. Dang, C. Davis, and B. Cooke. 2011. Mountain pine beetle host-range expansion threatens the boreal forest. Molecular Ecology 20:2157-2171.

Cumming, S., P. Burton, and B. Klinkenberg. 1996. Boreal mixedwood forests may have no "representative" areas: some implications for reserve design. Ecography 19:162-180.

Cumming, S., F. Schmiegelow, and P. Burton. 2000. Gap dynamics in boreal aspen stands: is the forest older than we think? Ecological Applications 10:744-759.

CWF. 2017. CWF Urges Immediate Action for Species at Risk on Prairie Grasslands. Canadian Wildlife Federation, press release Feb. 23, 2017, online: http://cwf-fcf.org/en/news-features/news-media/releases-1/.

Cyr, D., S. Gauthier, Y. Bergeron, and C. Carcaillet. 2009. Forest management is driving the eastern North American boreal forest outside its natural range of variability. Frontiers in Ecology and the Environment 7:519-524.

D'Orangeville, L., L. Duchesne, D. Houle, D Kneeshaw, and B. Côté. 2016. Northeastern North America as a potential refugium for boreal forests in a warming climate. Science 352:1452-1455.

Daily, G. and K. Ellison. 2002. The New Economy of Nature: The Quest to Make Conservation Profitable. Island Press, Washington, DC.

Daily, G., S. Polasky, J. Goldstein, P. Kareiva, and H. Mooney. 2009. Ecosystem services in decision making: time to deliver. Frontiers in Ecology and the Environment 7:21-28.

Daniels, L. and R. Gray. 2006. Disturbance regimes in coastal British Columbia. Journal of Ecosystems and Management 7:44-56.

Daoust, P.Y., E.L. Couture, T. Wimmer, and L. Bourque. 2018. Incident Report: North Atlantic Right Whale Mortality Event in the Gulf of St. Lawrence, 2017. Canadian Wildlife Health Cooperative and Fisheries and Oceans Canada, Ottawa, ON.

Dart, R. 2010. A grounded qualitative study of the meanings of effectiveness in Canadian 'results-focused' environmental organizations. Voluntas 21:202-219.

Dawe, K. and S. Boutin. 2016. Climate change is the primary driver of white-tailed deer (Odocoileus virginianus) range expansion at the northern extent of its range; land use is secondary. Ecology and Evolution 6:6435-6451.

Deguise, I. and J. Kerr. 2006. Protected areas and prospects for endangered species conservation in Canada. Conservation Biology 20:48-55.

Demulder, B. and W. Thorp. 2007. Integrated Landscape Management: Moving Forward. Alberta Chamber of Resources, Calgary, AB.

Denhoff, E. 2016. Setting Alberta on the Path to Caribou Recovery. Government of Alberta, Edmonton, AB.

Desrochers, R., J. Kerr, and D. Currie. 2011. How, and how much, natural cover loss increases species richness. Global Ecology and Biogeography 20:857-867.

Devictor, V. and A. Robert. 2009. Measuring community responses to large-scale disturbance in conservation biogeography. Diversity and Distributions 15:122-130.

Dextrase, A. and N. Mandrak. 2006. Impacts of alien invasive species on freshwater fauna at risk in Canada. Biological Invasions 8:13-24.

Diamond, J. 1975. The island dilemma: lessons of modern biogeographic studies for the design of natural reserves. Biological Conservation 7:129–146.

Diamond, J. 2005. Collapse: How Societies Choose to Fail or Succeed. Penguin, New York, NY.

Dicks, L., J. Walsh, and W. Sutherland. 2014. Organising evidence for environmental management decisions: a '4S' hierarchy. Trends in Ecology & Evolution 29:607-613.

Dilkina, B., R. Houtman, C. Gomes, C. Montgomery, and K. McKelvey. 2016. Trade-offs and efficiencies in optimal budget-constrained multispecies corridor networks. Conservation Biology 31:192-202.

Dinerstein, E., D. Olson, A. Joshi, C. Vynne, and N. Burgess. 2017. An ecoregion-based approach to protecting half the terrestrial realm. BioScience 67:534-545.

DLPC. 2006. Respect for the Land: the Dehcho Land-Use Plan. Dehcho Land-Use Planning Committee, Fort Providence, NT.

Doak, D., V. Bakker, B. Goldstein, and B. Hale. 2013. What is the future of conservation? Trends in Ecology & Evolution 29:77-81.

Dobak, W. 1996. Killing the Canadian Buffalo, 1821–1881. The Western Historical Quarterly 27:33-52.

Dobrowski, S. 2011. A climatic basis for microrefugia: the influence of terrain on climate. Global Change Biology 17:1022-1035.

Doerr, V., T. Barrett, and E. Doerr. 2011. Connectivity, dispersal behaviour and conservation under climate change: a response to Hodgson et al. Journal of Applied Ecology 48:143-147.

Donihee, J. 2000. The Evolution of Wildlife Law in Canada. Canadian Institute of Resources Law, Calgary, AB.

Dornelas, M., N. Gotelli, B. McGill, H. Shimadzu, and F. Moyes. 2014. Assemblage time series reveal biodiversity change but not systematic loss. Science 344:296-299.

Doughty, R. 1975. Feather Fashions and Bird Preservation: A Study in Nature Protection. University of California Press, Berkeley, CA.

Doyon, F., S. Yamasaki, and R. Duchesneau. 2008. The use of the natural range of variability for identifying biodiversity values at risk when implementing a forest management strategy. The Forestry Chronicle 84:316-329.

Drapeau, P., M. Villard, A. Leduc, and S. Hannon. 2016. Natural disturbance regimes as templates for the response of bird species assemblages to contemporary forest management. Diversity and Distributions 22:385-399.

Drever, C., G. Peterson, C. Messier, Y. Bergeron, and M. Flannigan. 2006. Can forest management based on natural disturbances maintain ecological resilience? Canadian Journal of Forest Research 36:2285-2299.

Driscoll, J., C. Robb, and K. Bodtker. 2009. Bycatch in Canada's Pacific Groundfish Bottom Trawl Fishery: Trends and Ecosystem Perspectives. Living Oceans Society, online report: https://livingoceans.org.

Drushka, K. 2003. Canada's Forests: A History. Forest History Society, Montreal, QC.

Dudley, N. 2008. Guidelines for Applying Protected Area Management Categories. International Union for the Conservation of Nature, Gland, Switzerland.

Duinker, P. 2001. Criteria and indicators of sustainable forest management in Canada: progress and problems in integrating science and politics at the local level. Pages 7-27 in Criteria and Indicators for Sustainable Forest Management at the Forest Management Unit Level, Proceedings of the European Forest Institute, No. 38, edited by A. Franc, Laroussinie O., and T. Karjalainen. European Forest Institute, Joensuu, Finland.

Duinker, P. 2011. Advancing the cause? Contributions of criteria and indicators to sustainable forest management in Canada. The Forestry Chronicle 87:488-493.

Duinker, P., E. Burbidge, S. Boardley, and L. Greig. 2012. Scientific dimensions of cumulative effects assessment: toward improvements in guidance for practice. Environmental Reviews 21:40-52.

Duinker, P. and L. Greig. 2006. The impotence of cumulative effects assessment in Canada: ailments and ideas for redeployment. Environmental Management 37:153-161.

Duinker, P. and L. Greig. 2007. Scenario analysis in environmental impact assessment: Improving explorations of the future. Environmental Impact Assessment Review 27:206-219.

Dunlop, M., H. Parris, P. Ryan, and F. Kroon. 2013. Climate-ready conservation objectives: a scoping study. National Climate Change Adaptation Research Facility, Gold Coast, Queensland.

Dyer, S., J. O'Neill, S. Wasel, and S. Boutin. 2002. Quantifying barrier effects of roads and seismic lines on movements of female woodland caribou in northeastern Alberta. Canadian Journal of Zoology 80:839-845.

Dyke, A. 2005. Late Quaternary vegetation history of northern North America based on pollen, macrofossil, and faunal remains. Géographie Physique et Quaternaire 59:211-262.

eBird 2023. 2022 Year in Review: eBird, Merlin, Macaulay Library, and Birds of the World. Online report: https://ebird.org/news/2022-year-in-review

EC. 1995. Canadian Biodiversity Strategy: Canada's Response to the Convention on Biological Diversity. Environment Canada, Ottawa, ON.

EC. 2004. An Invasive Alien Species Strategy for Canada. Environment Canada, Ottawa, ON.

EC. 2011. Scientific Assessment to Inform the Identification of Critical Habitat for Woodland Caribou (Rangifer tarandus caribou), Boreal Population, in Canada: 2011 Update. Environment Canada, Ottawa, ON.

EC. 2012a. The State of Canada's Birds – 2012. Environment Canada, Ottawa, ON.

EC. 2012b. Invasive Alien Species Partnership Program: 2005-2010 Report. Environment Canada, Ottawa, ON.

EC. 2012c. Recovery Strategy for the Woodland Caribou (Rangifer tarandus caribou), Boreal Population, in Canada. Environment Canada, Ottawa, ON.

EC. 2013. Action Plan for the Piping Plover (Charadrius melodus circumcinctus) in Ontario. Environment Canada, Ottawa, ON.

EC. 2015. Recovery Strategy for the Loggerhead Shrike, migrans subspecies (Lanius ludovicianus migrans), in Canada. Environment Canada, Ottawa, ON.

EC. 2016. Recovery Strategy for the Canada Warbler (Cardellina canadensis) in Canada. Species at Risk Act Recovery Strategy Series. Environment Canada, Ottawa, ON.

ECA. 1979. The Environmental Effects of Forestry Operations in Alberta: Report and Recommendations. Environmental Council of Alberta, Edmonton, AB.

ECCC. 2016a. Listing Policy for Terrestrial Species at Risk [Proposed]. Species at Risk Act: Policies and Guidelines Series. Environment and Climate Change Canada, Ottawa, ON.

ECCC. 2016b. Species at Risk Act Implementation Guidance for Recovery Practitioners: Critical Habitat Identification Toolbox. Version 2.3. Environment and Climate Change Canada, Ottawa, ON.

ECCC. 2016c. Range Plan Guidance for Woodland Caribou, Boreal Population. Species at Risk Act: Policies and Guidelines Series. Environment and Climate Change Canada, Ottawa, ON.

ECCC. 2017a. Species at Risk Act: Annual Report for 2016. Environment and Climate Change Canada, Ottawa, ON.

ECCC. 2017b. Action Plan for Multiple Species at Risk in Southwestern Saskatchewan: South of the Divide. Species at Risk Act Action Plan Series. Environment and Climate Change Canada, Ottawa, ON.

ECCC 2020. Policy on Survival and Recovery. Species at Risk Act: Policies and Guidelines Series. Government of Canada, Ottawa, ON.

ECCC. 2022. Canada's conserved areas. Environment and Climate Change Canada, Ottawa, ON. Available at: canada.ca/en/environment-climate-change/services/environmental-indicators/conserved-areas.html

ECCC. 2023. Climate Trends and Variations Bulletin: Annual 2022. Environment and Climate Change Canada, Ottawa, ON.

ECO. 2011. Ontario's Commercial Fisheries Policies. Engaging Solutions, ECO Annual Report, 2010/11. Environmental Commissioner of Ontario. The Queen's Printer for Ontario, Toronto, ON

Ecojustice. 2012a. Failure to Protect: Grading Canada's Species at Risk Laws. Ecojustice, Vancouver, BC.

Ecojustice. 2012b. Conservation Groups Take Environment Minister to Federal Court Over Failure to Protect Woodland Caribou, Again. Ecojustice, online press release: www.ecojustice.ca/.

Edmonds, E. 1988. Population status, distribution, and movements of woodland caribou in west central Alberta. Canadian Journal of Zoology 66:817-826.

Ehrenfeld, D. 2000. War and peace and conservation biology. Conservation Biology 14:105-112.

Ehrlich, P. 2000. Evolution of an advocate. Science 287:2159-2159.

Eigenbrod, F., S. Hecnar, and L. Fahrig. 2008. The relative effects of road traffic and forest cover on anuran populations. Biological Conservation 141:35-46.

Elgin, E., H. Tunna, and L. Jackson. 2014. First confirmed records of Prussian carp, Carassius gibelio (Bloch, 1782) in open waters of North America. Bioinvasions Records 3:275-282.

Elith, J. and J. Leathwick. 2009. Species distribution models: ecological explanation and prediction across space and time. Annual Review of Ecology, Evolution, and Systematics 40:677-697.

Enquist, C., S. Jackson, G. Garfin, F. Davis, and L. Gerber. 2017. Foundations of translational ecology. Frontiers in Ecology and the Environment 15:541-550.

Environics. 2012. Focus Canada 2012. Environics Institute, Toronto, ON.

Environics. 2013. People and their Natural Environment – A National Survey of Canadians. Environics Institute, Toronto, ON.

Ewers, R., S. Thorpe, and R. Didham. 2007. Synergistic interactions between edge and area effects in a heavily fragmented landscape. Ecology 88:96-106.

Fahrig, L. 2003. Effects of habitat fragmentation on biodiversity. Annual Review of Ecology, Evolution, and Systematics 34:487-515.

Fahrig, L. 2013. Rethinking patch size and isolation effects: the habitat amount hypothesis. Journal of Biogeography 40:1649-1663.

Failing, L. and R. Gregory. 2003. Ten common mistakes in designing biodiversity indicators for forest policy. Journal of Environmental Management 68:121-132.

Failing, L., R. Gregory, and M. Harstone. 2007. Integrating science and local knowledge in environmental risk management: a decision-focused approach. Ecological Economics 64:47-60.

Fall, A. and J. Fall. 2001. A domain-specific language for models of landscape dynamics. Ecological Modelling 141:1-18.

Fauchald, P. 2010. Predator–prey reversal: a possible mechanism for ecosystem hysteresis in the North Sea? Ecology 91:2191-2197.

FBC. 2014. Environmentally Significant Areas in Alberta: 2014 Update. Fiera Biological Consulting, online report: www.albertaparks.ca/media/5425575/2014-esa-final-report-april-2014.pdf.

FCC. 2014. Western Canada Wilderness Committee, David Suzuki Foundation, Greenpeace Canada, Sierra Club of British Columbia Foundation, and Wildsight v. Minister of Fisheries and Oceans and Minister of the Environment. Federal Court of Canada, Ottawa, ON.

Feldman, M., L. Imbeau, P. Marchand, M. Mazerolle, and M. Darveau. 2021. PLoS One 16:e0234587.

Feynman, R. 1985. Surely You're Joking, Mr. Feynman! Bantam Books, New York, NY.

Fieberg, J. and S. Ellner. 2000. When is it meaningful to estimate an extinction probability? Ecology 81:2040-2047.

Fink, D., T. Auer, A. Johston, V. Ruiz-Gutierrez, and M. Hochachka. 2020. Modeling avian full annual cycle distribution and population trends with citizen science data. Ecological Applications 30:e02056.

Findlay, C., S. Elgie, B. Giles, and L. Burr. 2009. Species listing under Canada's species at risk act. Conservation Biology 23:1609-1617.

Fisher, J. and A. Burton. 2018. Wildlife winners and losers in an oil sands landscape. Frontiers in Ecology and the Environment 16:323-328.

Fisher, R. and E. Bayne. 2014. Burrowing Owl Climate Change Adaptation Plan for Alberta. Alberta Biodiversity Monitoring Institute, Edmonton, AB.

Fisher, R., S. Muszala, M. Verteinstein, P. Lawrence, and C. Xu. 2015a. Taking off the training wheels: the properties of a dynamic vegetation model without climate envelopes, CLM4. 5 (ED). Geoscientific Model Development 8:3593–3619.

Fisher, R., T. Wellicome, E. Bayne, R. Poulin, and L. Todd. 2015b. Extreme precipitation reduces reproductive output of an endangered raptor. Journal of Applied Ecology 52:1500-1508.

Fithian, W., J. Elith, T. Hastle, and D. Keith. 2015. Bias correction in species distribution models: pooling survey and collection data for multiple species. Methods in Ecology and Evolution 6:424-438.

Flannigan, M., B. Wotton, G. Marshall, W. De Groot, and. Johnston. 2016. Fuel moisture sensitivity to temperature and precipitation: climate change implications. Climatic Change 134:59-71.

Flather, C., G. Hayward, S. Beissinger, and P. Stephens. 2011. Minimum viable populations: is there a 'magic number' for conservation practitioners? Trends in Ecology & Evolution 26:307-316.

FOC. 2011. Recovery Strategy for the Northern and Southern Resident Killer Whales (Orcinus orca) in Canada. Fisheries and Oceans Canada, Ottawa, ON.

FOC. 2017a. Action Plan for the Northern and Southern Resident Killer Whale (Orcinus orca) in Canada. Fisheries and Oceans Canada, Ottawa, ON.

FOC. 2017b. Northern (NAFO Divs. 2J3KL) Cod Stock Update. Fisheries and Oceans Canada, Ottawa, ON.

Forbes, S. 2011. Science and policy: valuing framing, language and listening. Botanical Journal of the Linnean Society 166:217-226.

Foster, J. 1978. Working for Wildlife. University of Toronto Press, Toronto, ON.

FPB. 2008. Provincial Land Use Planning: Which Way from Here? Special Report 34. Forest Practices Board, Victoria, BC.

FPTMOA. 2021. The Guelph Statement: A Vision to 2028. Federal, Provincial and Territorial Ministers of Agriculture. Available at: https://agriculture.canada.ca/en/department/initiatives/meetings-ministers/guelph-statement

Francis, S. and J. Hamm. 2011. Looking forward: using scenario modeling to support regional land use planning in Northern Yukon, Canada. Ecology and Society 16:18.

Franks, D., D. Brereton, and C. Moran. 2013. The cumulative dimensions of impact in resource regions. Resources Policy 38:640-647.

Fraser, D. 2008. How well can captive breeding programs conserve biodiversity? A review of salmonids. Evolutionary Applications 1:535-586.

Fraser, D., L. Weir, L. Bernatchez, M. Hansen, and E. Taylor. 2011. Extent and scale of local adaptation in salmonid fishes: review and meta-analysis. Heredity 106:404-420.

Freeman, M. and G. Wenzel. 2006. The nature and significance of polar bear conservation hunting in the Canadian Arctic. Arctic 59:21-30.

Freese, C., S. Fuhlendorf, and K. Kunkel. 2014. A management framework for the transition from livestock production toward biodiversity conservation on great plains rangelands. Ecological Restoration 32:358-368.

Frick, W., J. Pollock, A. Hicks, K. Langwig, and D. Reynolds. 2010. An emerging disease causes regional population collapse of a common North American bat species. Science 329:679-682.

FSC. 2020. FSC Facts and Figures. Forest Stewardship Council, Feb. 17, 2020 online report: https://fsc.org/sites/default/files/2020-02/Facts_and_Figures_2020-02-17.pdf

Fuhlendorf, S. and D. Engle. 2001. Restoring heterogeneity on rangelands: ecosystem management based on evolutionary grazing patterns. BioScience 51:625-632.

Fuhlendorf, S., D. Engle, R. Elmore, R. Limb, and T. Bidwell. 2012. Conservation of pattern and process: developing an alternative paradigm of rangeland management. Rangeland Ecology & Management 65:579-589.

Fuhlendorf, S., W. Harrell, D. Engle, R. Hamilton, and C. Davis. 2006. Should heterogeneity be the basis for conservation? Grassland bird response to fire and grazing. Ecological Applications 16:1706-1716.

Fuller, W. and S. Novakowski. 1955. Wolf Control Operations, Wood Buffalo National Park, 1951–1952. Wildlife Management Bulletin Number 11, Canadian Wildlife Service , Ottawa, ON.

Gadgil, M., F. Berkes, and C. Folke. 1993. Indigenous knowledge for biodiversity conservation. Ambio 22:151-156.

Gagnon, C. and D. Berteaux. 2009. Integrating traditional ecological knowledge and ecological science: a question of scale. Ecology and Society 14:19.

Gallagher, R., R. Makinson, P. Hogbin, and N. Hancock. 2015. Assisted colonization as a climate change adaptation tool. Austral Ecology 40:12-20.

Game, E. and H.S. Grantham. 2008. Marxan User Manual: For Marxan Version 1.8.10. University of Queensland, St. Lucia, Queensland, Australia.

Game, E., P. Kareiva, and H. Possingham. 2013. Six common mistakes in conservation priority setting. Conservation Biology 27:480-485.

Game, E., G. Lipsett-Moore, E. Saxon, N. Peterson, and S. Sheppards. 2011. Incorporating climate change adaptation into national conservation assessments. Global Change Biology 17:3150-3160.

Game, E., M. Schwartz, and A. Knight. 2015. Policy relevant conservation science. Conservation Letters 8:309-311.

Gardmark, A., M. Casini, M. Huss, A. van Leeuwen, and J. Hjelm. 2015. Regime shifts in exploited marine food webs: detecting mechanisms underlying alternative stable states using size-structured community dynamics theory. Philosophical Transactions of the Royal Society of London B: Biological Sciences 370:20130262.

Gaston, K. 2009. Geographic range limits: achieving synthesis. Proceedings of the Royal Society of London B: Biological Sciences 276:1395-1406.

Gauthier, S., T. Kuuluvainen, E. Macdonald, E. Shorohova, and A. Shvidenko. 2023. Ecosystem management of the boreal forest in the era of global change. Pages 3-49 in Boreal Forests in the Face of Climate Change, edited by M. Girona, S. Gauthier, H. Morin, and Y. Bergeron. Advances in Global Change Research, Volume 74.

Gehlhausen, S., M. Schwartz, and C. Augspurger. 2000. Vegetation and microclimatic edge effects in two mixed-mesophytic forest fragments. Plant Ecology 147:21-35.

Geist, V. 1988. How markets in wildlife meat and parts, and the sale of hunting privileges, jeopardize wildlife conservation. Conservation Biology 2:15-26.

George, A., B. Kuhajda, J. Williams, M. Cantrell, and P. Rakes. 2009. Guidelines for propagation and translocation for freshwater fish conservation. Fisheries 34:529-545.

Gerber, L. 2016. Conservation triage or injurious neglect in endangered species recovery. Proceedings of the National Academy of Sciences 113:3563-3566.

Gibbs, K., R. Mackey, and D. Currie. 2009. Human land use, agriculture, pesticides and losses of imperiled species. Diversity and Distributions 15:242-253.

Gibson, S., R. van der Marel, and B. Starzomski. 2009. Climate change and conservation of leading-edge peripheral populations. Conservation Biology 23:1369-1373.

Glac, K. 2009. Understanding socially responsible investing: The effect of decision frames and trade-off options. Journal of Business Ethics 87:41-55.

Glick, P., H. Chmura, and B. Stein. 2011. Moving the Conservation Goalposts: A Review of Climate Change Adaptation Literature. National Wildlife Federation, Merrifield, Virginia.

GOA. 2008. Land-use Framework. Government of Alberta, Edmonton, AB.

GOA. 2011. A Woodland Caribou Policy for Alberta. Government of Alberta, Edmonton, AB.

GOA. 2012. Lower Athabasca Regional Plan 2012–2022. Government of Alberta, Edmonton, AB.

GOA. 2015. Surface Water Quality Management Framework for the Lower Athabasca River. Government of Alberta, Edmonton, AB.

GOA. 2016. Environmental Protection and Enhancement Act: Conservation and Reclamation Regulation. Government of Alberta, Edmonton, AB.

GOA. 2017a. Draft Provincial Woodland Caribou Range Plan. Government of Alberta, Edmonton, AB.

GOA. 2017b. North Central Native Trout Recovery Program. Government of Alberta, Edmonton, AB.

GOA. 2018a. Lower Athabasca Regional Plan—Conservation Areas: Q & A. Government of Alberta, Edmonton, AB.

GOA. 2018b. Northern Pike and Walleye Fisheries Management Frameworks: Summary of Survey Feedback. Government of Alberta, Edmonton, AB.

GOBC. 1994. Forest Practices Code of British Columbia Act. Government of BC, Victoria, BC.

GOBC. 2013. Guideline for the Selection of Valued Components and Assessment of Potential Effects. Government of BC, Environmental Assessment Office, Victoria, BC.

GOBC. 2014. Mount Polley Tailings Pond Situation Update. Government of BC, Victoria, BC.

GOC. 2000. Canada National Parks Act. Government of Canada, Ottawa, ON.

GOC. 2002. Species at Risk Act. Government of Canada, Ottawa, ON.

GOC. 2011a. Ballast Water Control and Management Regulations. Government of Canada, Ottawa, ON.

GOC. 2011b. National Framework for Canada's Network of Marine Protected Areas. Government of Canada, Ottawa, ON.

GOC. 2015. Canada Wildlife Act. Government of Canada, Ottawa, ON.

GOC. 2018. Organic Production Systems: General Principles and Management Standards. Government of Canada, Ottawa, ON.

GOC. 2022a. A Plan to Grow Our Economy and Mike Life More Affordable: 2022 Budget. Government of Canada, Ottawa, ON.

GOC 2022b. News Release: Government of Canada recognizing federal land and water to contribute to 30 by 30 nature conservation goals. Government of Canada, Ottawa, ON.

Godefroid, S., C. Piazza, G. Rossi, S. Buord, and A. Stevens. 2011. How successful are plant species reintroductions? Biological Conservation 144:672-682.

Gomez-Baggethun, E. and M. Ruiz-Perez. 2011. Economic valuation and the commodification of ecosystem services. Progress in Physical Geography 35:613-628.

GONWT. 2016. Final Written Submission for the 2016 Bathurst Caribou Herd Proceeding. Government of the Northwest Territories, online report: www.wrrb.ca/.

GOO. 1994. Crown Forest Sustainability Act. Government of Ontario, Toronto, ON.

Good, K. and S. Michalsky. 2008. Summary of Canadian Experience With Conservation Easements and Their Potential Application to Agri-Environmental Policy. Agriculture and Agri-Food Canada, Ottawa, ON.

GOQ. 2015. The Plan Nord: 2015-2020 Action Plan. Government of Quebec, Quebec City, QC.

Grandy, J. 2013. Environmental Charities in Canada. Charity Intelligence Canada, Toronto, ON.

Grantham, H., M. Bode, E. McDonald-Madden, E. Game, and A. Knight. 2010. Effective conservation planning requires learning and adaptation. Frontiers in Ecology and the Environment 8:431-437.

Gray, L. and A. Hamann. 2013. Tracking suitable habitat for tree populations under climate change in western North America. Climatic Change 117:289-303.

Gray, S., A. Chan, D. Clark, and R. Jordan. 2012. Modeling the integration of stakeholder knowledge in social-ecological decision-making: benefits and limitations to knowledge diversity. Ecological Modelling 229:88-96.

Green, D. 2005. Designatable units for status assessment of endangered species. Conservation Biology 19:1813-1820.

Gregg, A. and M. Posner. 1990. The Big Picture: What Canadians Think About Almost Every-thing. Macfarlane Walter & Ross, Toronto, ON.

Gregory, R., J. Arvai, and L. Gerber. 2013. Structuring decisions for managing threatened and endangered species in a changing climate. Conservation Biology 27:1212-1221.

Gregory, R., L. Failing, M. Harstone, G. Long, T. McDaniels, and D. Ohlson. 2012. Structured Decision Making: A Practical Guide to Environmental Management Choices. Wiley-Blackwell, West Sussex, UK.

Gregory, R., L. Failing, and P. Higgins. 2006. Adaptive management and environmental decision making: a case study application to water use planning. Ecological Economics 58:434-447.

Gregory, R. and G. Long. 2009. Using structured decision making to help implement a precautionary approach to endangered species management. Risk Analysis 29:518-532.

Grondin, P., S. Gauthier, V. Poirier, P. Tardif, and Y. Boucher. 2018. Have some landscapes in the eastern Canadian boreal forest moved beyond their natural range of variability? Forest Ecosystems 5:30.

Groves, C., E. Game, M. Anderson, M. Cross, and C. Enquist. 2012. Incorporating climate change into systematic conservation planning. Biodiversity and Conservation 21:1651-1671.

Grumbine, R. 1994. What is ecosystem management? Conservation Biology 8:27-38.

Gueterbock, R. 2004. Greenpeace campaign case study—StopEsso. Journal of Consumer Behaviour 3:265-271.

Gunderson, L. 2000. Ecological resilience—in theory and application. Annual Review of Ecology and Systematics 31:425-439.

Gunn, J. and B. Noble. 2009. Integrating cumulative effects in regional strategic environmental assessment frameworks: lessons from practice. Journal of Environmental Assessment Policy and Management 11:267-290.

Gurd, D., T. Nudds, and D. Rivard. 2001. Conservation of mammals in eastern North American wildlife reserves: how small is too small? Conservation Biology 15:1355-1363.

Gurmendi, A. 2004. Minerals Yearbook, The Mineral Industry of Canada. Bureau of Mines, US Geological Survey.

Haddad, N., A. Gonzalez, L. Brudvig, M. Burt, and D. Levey. 2017. Experimental evidence does not support the Habitat Amount Hypothesis. Ecography 40:48-55.

Haddad, N., J. Haarstad, and D. Tilman. 2000. The effects of long-term nitrogen loading on grassland insect communities. Oecologia 124:73-84.

Hagerman, S., H. Dowlatabadi, K. Chan, and T. Satterfield. 2010a. Integrative propositions for adapting conservation policy to the impacts of climate change. Global Environmental Change 20:351-362.

Hagerman, S., H. Dowlatabadi, T. Satterfield, and T. McDaniels. 2010b. Expert views on biodiversity conservation in an era of climate change. Global Environmental Change 20:192-207.

Hagerman, S. and T. Satterfield. 2013. Entangled judgments: expert preferences for adapting biodiversity conservation to climate change. Journal of Environmental Management 129:555-563.

Hagerman, S. and T. Satterfield. 2014. Agreed but not preferred: expert views on taboo options for biodiversity conservation, given climate change. Ecological Applications 24:548-559.

Hall, M., M. de Wit, D. Lasby, D. McIver, and T. Evers. 2005. Cornerstones of Community: Highlights of the National Survey of Nonprofit and Voluntary Organizations. Report 61-533-XIE. Statistics Canada, Ottawa, ON.

Hallett, L., T. Morelli, L. Gerber, M. Moritz, and M. Schwartz. 2017. Navigating translational ecology: creating opportunities for scientist participation. Frontiers in Ecology and the Environment 15:578-586.

Hamann, A. and S. Aitken. 2013. Conservation planning under climate change: accounting for adaptive potential and migration capacity in species distribution models. Diversity and Distributions 19:268-280.

Hamann, A. and T. Wang. 2006. Potential effects of climate change on ecosystem and tree species distribution in British Columbia. Ecology 87:2773-2786.

Hamann, A., D. Roberts, Q. Barber, C. Carroll, and S. Nielsen. 2015. Velocity of climate change algorithms for guiding conservation and management. Global Change Biology 21:997-1004.

Hampe, A. and R. Petit. 2005. Conserving biodiversity under climate change: the rear edge matters. Ecology Letters 8:461-467.

Hancock, N. and R. Gallagher. 2014. How ready are we to move species threatened from climate change? Insights into the assisted colonization debate from Australia. Austral ecology 39:830-838.

Hannon, S. and P. Drapeau. 2005. Bird responses to burning and logging in the boreal forest of Canada. Studies in Avian Biology 30:97.

Hannon, S. and P. Drapeau. 2005. Bird responses to burns and clear cuts in the boreal forest of Canada. Studies in Avian Biology 30:97-115.

Hansen, A. 2010. Environment, Media and Communication. Routledge, New York, NY.

Hansen, J., P. Kharecha, M. Sato, V. Masson-Delmotte, and F. Ackerman. 2013. Assessing "dangerous climate change": required reduction of carbon emissions to protect young people, future generations and nature. PloS One 8:e81648.

Hanski, I. 1998. Metapopulation dynamics. Nature 396:41-49.

Hanski, I. 2015. Habitat fragmentation and species richness. Journal of Biogeography 42:989-993.

Hardin, G. 1968. The tragedy of the commons. Science 162:1243-1248.

Hargreaves, A., K. Samis, and C. Eckert. 2014. Are species' range limits simply niche limits writ large? A review of transplant experiments beyond the range. The American Naturalist 183:157-173.

Harriman, J. and B. Noble. 2008. Characterizing project and strategic approaches to regional cumulative effects assessment in Canada. Journal of Environmental Assessment Policy and Management 10:25-50.

Harris, J., R. Hobbs, E. Higgs, and J. Aronson. 2006. Ecological restoration and global climate change. Restoration Ecology 14:170-176.

Hauer, G., S. Cumming, F. Schmiegelow, W. Adamowicz, and M. Weber. 2010. Tradeoffs between forestry resource and conservation values under alternate policy regimes: A spatial analysis of the western Canadian boreal plains. Ecological Modelling 221:2590-2603.

Hayes, D., C. Ferreri, and W. Taylor. 1996. Linking fish habitat to their population dynamics. Canadian Journal of Fisheries and Aquatic Sciences 53:383-390.

Hayhoe, K. and J. Schwartz. 2017. The roots of science denial. Scientific American 317:66-68.

Hebblewhite, M. 2017. Billion dollar boreal woodland caribou and the biodiversity impacts of the global oil and gas industry. Biological Conservation 206:102-111.

Hebblewhite, M. and D. Haydon. 2010. Distinguishing technology from biology: a critical review of the use of GPS telemetry data in ecology. Philosophical Transactions of the Royal Society of London B: Biological Sciences 365:2303-2312.

Hebblewhite, M., C. White, C. Nietvelt, J. McKenzie, and T. Hurd. 2005. Human activity mediates a trophic cascade caused by wolves. Ecology 86:2135-2144.

Heller, N. and E. Zavaleta. 2009. Biodiversity management in the face of climate change: a review of 22 years of recommendations. Biological Conservation 142:14-32.

Henderson, A., M. Reed, and S. Davis. 2014. Voluntary stewardship and the Canadian Species at Risk Act: exploring rancher willingness to support species at risk in the Canadian prairies. Human Dimensions of Wildlife 19:17-32.

Herman, D. 2003. The hunter's aim: the cultural politics of American sport hunters, 1880–1910. Journal of Leisure Research 35:455.

Herraro, S. 2003. Canada's experimental reintroduction of swift foxes into an altered ecosystem. Pages 33-38 in The Swift Fox: Ecology and Conservation of Swift Foxes in a Changing World, edited by M. Sovada and L. Carbyn. Canadian Plans Research Center, University of Regina, Regina, SK.

Herrero, S., C. Schroeder, and M. Scott-Brown. 1986. Are Canadian foxes swift enough? Biological Conservation 36:159-167.

Hervieux, D., J. Edmonds, R. Bonar, and J. McCammon. 1996. Successful and unsuccessful attempts to resolve caribou management and timber harvesting issues in west central Alberta. Rangifer 16:185-190.

Hervieux, D., M. Hebblewhite, N. DeCesare, M. Russell, and K. Smith. 2013. Widespread declines in woodland caribou (Rangifer tarandus caribou) continue in Alberta. Canadian Journal of Zoology 91:872-882.

Hervieux, D., M. Hebblewhite, D. Stepnisky, M. Bacon, and S. Boutin. 2014. Managing wolves (Canis lupus) to recover threatened woodland caribou (Rangifer tarandus caribou) in Alberta. Canadian Journal of Zoology 92:1029-1037.

Hewitt, N., N. Klenk, A. Smith, D. Bazely, and N. Yan. 2011. Taking stock of the assisted migration debate. Biological Conservation 144:2560-2572.

Higgins, J., T. Ricketts, J. Parrish, E. Dinerstein, and G. Powell. 2004. Beyond Noah: saving species is not enough. Conservation Biology 18:1672-1673.

Higgs, E. 2017. Novel and designed ecosystems. Restoration Ecology 25:8-13.

Hillary, R. 2004. Environmental management systems and the smaller enterprise. Journal of Cleaner Production 12:561-569.

HilleRisLambers, J., M. Harsch, A. Ettinger, K. Ford, and E. Theobald. 2013. How will biotic interactions influence climate change-induced range shifts? Annals of the New York Academy of Sciences 1297:112-125.

Hilliard, C., N. Scott, A. Lessa, and S. Reedyk. 2002. Agricultural Best Management Practices for the Canadian Prairies: A Review of Literature. Agriculture and Agri-Food Canada, Ottawa, ON.

Hilson, G. and B. Murck. 2000. Sustainable development in the mining industry: clarifying the corporate perspective. Resources Policy 26:227-238.

Hilty, J., W. Lidicker, and A. Merenlender. 2006. Corridor Ecology: The Science and Practice of Linking Landscapes for Biodiversity Conservation. Island Press, Washington, DC.

Hobbs, R. and V. Cramer. 2008. Restoration ecology: interventionist approaches for restoring and maintaining ecosystem function in the face of rapid environmental change. Annual Review of Environment and Resources 33:39-61.

Hobson, K. and E. Bayne. 2000. Breeding bird communities in boreal forest of western Canada: consequences of “unmixing” the mixedwoods. The Condor 102:759-769.

Hobson, K., E. Bayne, and S. Van Wilgenburg. 2002. Large-scale conversion of forest to agriculture in the boreal plains of Saskatchewan. Conservation Biology 16:1530-1541.

Hockings, M. 2003. Systems for assessing the effectiveness of management in protected areas. BioScience 53:823-832.

Hodge, G., H. Hall, and I. Robinson. 2016. Planning Canadian Regions, Second Edition. UBC Press, Vancouver, BC.

Hodgson, J., C. Thomas, B. Wintle, and A. Moilanen. 2009. Climate change, connectivity and conservation decision making: back to basics. Journal of Applied Ecology 46:964-969.

Hof, C., I. Levinsky, M. Araujo, and C. Rahbek. 2011. Rethinking species’ ability to cope with rapid climate change. Global Change Biology 17:2987-2990.

Hogg, E. and P. Bernier. 2005. Climate change impacts on drought-prone forests in western Canada. The Forestry Chronicle 81:675-682.

Holroyd, G. and D. Bird. 2012. Lessons learned during the recovery of the peregrine falcon in Canada. Canadian Wildlife Biology Management 1:3-20.

Honea, J., M. McClure, J. Jorgensen, and M. Scheuerell. 2016. Assessing freshwater life-stage vulnerability of an endangered Chinook salmon population to climate change influences on stream habitat. Climate Research 71:127-137.

Hood, G. and D. Larson. 2015. Ecological engineering and aquatic connectivity: a new perspective from beaver-modified wetlands. Freshwater Biology 60:198-208.

Horton, C., T. Peterson, P. Banerjee, and M. Peterson. 2016. Credibility and advocacy in conservation science. Conservation Biology 30:23-32.

Houde, N. 2007. The six faces of traditional ecological knowledge: challenges and opportunities for Canadian co-management arrangements. Ecology and Society 12:34.

Huddart-Kennedy, E., T. Beckley, B. McFarlane, and S. Nadeau. 2009. Rural-urban differences in environmental concern in Canada. Rural Sociology 74:309-329.

Hummel, M. 1989. The upshot. Pages 267-274 in Endangered Spaces: The Future for Canada's Wilderness, edited by M. Hummel. Key Porter Books, Toronto, ON.

Hunt, L. and N. Lester. 2009. The effect of forestry roads on access to remote fishing lakes in northern Ontario, Canada. North American Journal of Fisheries Management 29:586-597.

Hunter, M.L. and A. Calhoun 1996. A triad approach to land-use allocation. Pages 477-491 in Biodiversity in Managed Landscapes, edited by R.C. Szaro and D. Johnstone. Oxford University Press, Oxford, UK.

Hunter, M., G. Jacobson, and T. Webb. 1988. Paleoecology and the coarse-filter approach to maintaining biological diversity. Conservation Biology 2:375-385.

IEEIRP. 2015. Report on Mount Polley Tailings Storage Facility Breach. Government of BC, Victoria, BC.

IKEA. 2014. Sustainability Report FY14. IKEA Online report: http://www.ikea.com/ms/en_ca/this-is-ikea/people-and-planet/index.html.

Illical, M. and K. Harrison. 2007. Protecting endangered species in the US and Canada: the role of negative lesson drawing. Canadian Journal of Political Science 40:367-394.

INAC. 2007. Mine Site Reclamation Guidelines for the Northwest Territories. Indian and Northern Affairs Canada, Yellowknife, NWT.

Ingram, H. and L. Fraser 2006. Path dependency and adroit innovation: the case of California water. Pages 78-109 in Punctuated Equilibrium and the Dynamics of U.S. Environmental Policy, edited by R. Repetto. Yale University Press, New Haven, CT.

IPSOS. 2017. Nearly Nine in 10 Canadians (89%) Consider Preventing the Extinction of Wild Plants and Animals Important. IPSOS press release: www.ipsos.com/en-ca.

Isaac, N., S. Turvey, B. Collen, C. Waterman, and J. Baillie. 2007. Mammals on the EDGE: conservation priorities based on threat and phylogeny. PloS One 2:1-e296.

IUCN. 2013. Guidelines for Reintroductions and Other Conservation Translocations. Version 1.0. International Union for Conservation of Nature, Gland, Switzerland.

Jachowski, D. and D. Kesler. 2009. Allowing extinction: should we let species go? Trends in Ecology & Evolution 24:180.

Jackson, L., A. Trebitz, and K. Cottingham. 2000. An introduction to the practice of ecological modeling. BioScience 50:694-706.

Jacobson, S. and M. Duff. 1998. Training idiot savants: the lack of human dimensions in conservation biology. Conservation Biology 12:263-267.

Jacques, P., R. Dunlap, and M. Freeman. 2008. The organisation of denial: conservative think tanks and environmental scepticism. Environmental Politics 17:349-385.

Jamieson, I. 2011. Founder effects, inbreeding, and loss of genetic diversity in four avian reintroduction programs. Conservation Biology 25:115-123.

Javorek, S. and M. Grant. 2011. Trends in Wildlife Habitat Capacity on Agricultural Land in Canada, 1986–2006. Canadian Biodiversity: Ecosystem Status and Trends 2010, Technical Thematic Report No. 14. Canadian Council of Resource Ministers. Ottawa, ON.

Jia, G., H. Epstein, and D. Walker. 2009. Vegetation greening in the Canadian Arctic related to decadal warming. Journal of Environmental Monitoring 11:2231-2238.

Jiang, J., H. Su, C. Zhai, V. Perun, and A. Del Genio. 2012. Evaluation of cloud and water vapor simulations in CMIP5 climate models using NASA "A-Train" satellite observations. Journal of Geophysical Research: Atmospheres 117:1-24.

Johannes, R. 2002. Did indigenous conservation ethics exist? SPC Traditional Marine Resource Management and Knowledge Information Bulletin 14:3-7.

Johnson, C., M. Boyce, R. Case, H. Cluff, and R. Gau. 2005. Cumulative effects of human developments on arctic wildlife. Wildlife Monographs 160:1-36.

Johnson, E. and K. Miyanishi. 2008. Creating new landscapes and ecosystems. Annals of the New York Academy of Sciences 1134:120-145.

Johnson, E., K. Miyanishi, and J. Weir. 1998. Wildfires in the western Canadian boreal forest: landscape patterns and ecosystem management. Journal of Vegetation Science 9:603-610.

Johnston, M., T. Williamson, A. Munson, A. Ogden, and M. Moroni. 2010. Climate Change and Forest Management in Canada: Impacts, Adaptive Capacity and Adaptation Options. A State of Knowledge Report. Sustainable Forest Management Network, Edmonton, Alberta.

Jones, F. 2016. Cumulative effects assessment: theoretical underpinnings and big problems. Environmental Reviews 24:187-204.

Joseph, L., R. Maloney, and H. Possingham. 2009. Optimal allocation of resources among threatened species: a project prioritization protocol. Conservation Biology 23:328-338.

Jost, L. 2007. Partitioning diversity into independent alpha and beta components. Ecology 88:2427-2439.

Kahneman, D. 2011. Thinking, Fast and Slow. Farrar, Straus and Giroux, New York, NY.

Kareiva, P. and M. Marvier. 2012. What is conservation science? BioScience 62:962-969.

Karr, J. 2006. When government ignores science, scientists should speak up. BioScience 56:287-288.

Kaufman, S., E. Snucins, J. Gunn, and W. Selinger. 2009. Impacts of road access on lake trout (Salvelinus namaycush) populations: regional scale effects of overexploitation and the introduction of smallmouth bass (Micropterus dolomieu). Canadian Journal of Fisheries and Aquatic Sciences 66:212-223.

Keane, R., P. Hessburg, P. Landres, and F. Swanson. 2009. The use of historical range and variability (HRV) in landscape management. Forest Ecology and Management 258:1025-1037.

Keller, W., N. Yan, J.M. Gunn, and J. Heneberry. 2007. Recovery of acidified lakes: lessons from Sudbury, Ontario, Canada. Water, Air, & Soil Pollution: Focus 7:317-322.

Kelly, A. and M. Goulden. 2008. Rapid shifts in plant distribution with recent climate change. Proceedings of the National Academy of Sciences 105:11823-11826.

Kelly, R., A. Erickson, L. Mease, W. Battista, and J. Kittinger. 2015. Embracing thresholds for better environmental management. Philosophical Transactions of the Royal Society of London B: Biological Sciences 370:20130276.

Kennedy, E., T. Beckley, B. McFarlane, and S. Nadeau. 2009. Why we don't "walk the talk": understanding the environmental values/behaviour gap in Canada. Human Ecology Review 16:151.

Kennedy, P. and J. Donihee. 2006. Wildlife and the Canadian Constitution. Canadian Institute of Resources Law, Calgary, AB.

Kennett, S. 1999. Towards a New Paradigm for Cumulative Effects Management. Occasional Paper #8. Canadian Institute of Resources Law, Calgary, AB.

Kennett, S. 2002. Integrated Resource Management in Alberta: Past, Present and Benchmarks for the Future. Occasional Paper #11. Canadian Institute of Resources Law, Calgary, AB.

Kennett, S. 2006a. From Science-Based Thresholds to Regulatory Limits: Implementation Issues for Cumulative Effects Management. Canadian Institute of Resources Law, Calgary, AB.

Kennett, S. 2006b. Integrated Landscape Management in Canada: Getting from Here to There. Canadian Institute of Resources Law, Calgary, AB.

Kenney, A., S. Elgie, D. Sawyer, and C. Wichtendahl. 2011. Advancing the Economics of Ecosystems and Biodiversity in Canada: A Survey of Economic Instruments for the Conservation and Protection of Biodiversity. Environment Canada, Ottawa, ON.

Kerr, D., D. Holdsworth, S. Laskin, and G. Matthews. 1990. Historical Atlas of Canada. Volume III: Addressing the 20th Century 1891–1961. University of Toronto Press, Toronto, ON.

Kerr, J. and J. Cihlar. 2004. Patterns and causes of species endangerment in Canada. Ecological Applications 14:743-753.

KIMC. 1995. Kamloops Land and Resource Management Plan. Kamloops Interagency Management Committee, Kamloops, BC.

Kissling, W., R. Field, H. Korntheuer, U. Heyder, and K. Bohning-Gaese. 2010. Woody plants and the prediction of climate-change impacts on bird diversity. Philosophical Transactions of the Royal Society of London B: Biological Sciences 365:2035-2045.

Klenk, N. 2015. The development of assisted migration policy in Canada: an analysis of the politics of composing future forests. Land Use Policy 44:101-109.

Klijn, E. 2008. Complexity theory and public administration: what's new; key concepts in complexity theory compared to their counterparts in public administration. Public Management Review 10:299-317.

Kneeshaw, D. and S. Gauthier. 2003. Old growth in the boreal forest: a dynamic perspective at the stand and landscape level. Environmental Reviews 11:S99-S114.

Knight, A., R. Cowling, and B. Campbell. 2006. An operational model for implementing conservation action. Conservation Biology 20:408-419.

Knight, A., R. Cowling, M. Rouget, A. Balmford, and A. Lombard. 2008. Knowing but not doing: selecting priority conservation areas and the research–implementation gap. Conservation Biology 22:610-617.

Koen, E., J. Bowman, C. Sadowski, and A. Walpole. 2014a. Landscape connectivity for wildlife: development and validation of multispecies linkage maps. Methods in Ecology and Evolution 5:626-633.

Koen, E., J. Bowman, D. Murray, and P. Wilson. 2014b. Climate change reduces genetic diversity of Canada lynx at the trailing range edge. Ecography 37:754-762.

Krause, V. 2013. New U.S. funding for the war on Canadian oil. Financial Post, online news article: https://business.financialpost.com.

Kremer, A., O. Ronce, J. Robledo-Arnuncio, F. Guillaume, and G. Bohrer. 2012. Long-distance gene flow and adaptation of forest trees to rapid climate change. Ecology Letters 15:378-392.

Kreutzweiser, D., F. Beall, K. Webster, D. Thompson, and I. Creed. 2013. Impacts and prognosis of natural resource development on aquatic biodiversity in Canada's boreal zone. Environmental Reviews 21:227-259.

Kristensen, S., B. Noble, and R. Patrick. 2013. Capacity for watershed cumulative effects assessment and management: lessons from the lower Fraser River Basin, Canada. Environmental Management 52:360-373.

Kukkala, A. and A. Moilanen. 2013. Core concepts of spatial prioritisation in systematic conservation planning. Biological Reviews 88:443-464.

Kunreuther, H., G. Heal, M. Allen, O. Edenhofer, and C. Field. 2013. Risk management and climate change. Nature Climate Change 3:447-450.

Kuuluvainen, T. and S. Gauthier. 2018. Young and old forest in the boreal: critical stages of ecosystem dynamics and management under global change. Forest Ecosystems 5:26.

Kuuluvainen, T. and R. Grenfell. 2012. Natural disturbance emulation in boreal forest ecosystem management—theories, strategies, and a comparison with conventional even-aged management. Canadian Journal of Forest Research 42:1185-1203.

Kuussaari, M., R. Bommarco, R. Heikkinen, A. Helm, and J. Krauss. 2009. Extinction debt: a challenge for biodiversity conservation. Trends in Ecology & Evolution 24:564-571.

Lach, D., P. List, B. Steel, and B. Shindler. 2003. Advocacy and credibility of ecological scientists in resource decision making: a regional study. Bio-Science 53:170-178.

Lackey, R. 2006. Axioms of ecological policy. Fisheries 31:286-290.

Lackey, R. 2007. Science, scientists, and policy advocacy. Conservation Biology 21:12-17.

Lagios, E., K. Robbins, J. Lapierre, J. Steiner, and T. Imlay. 2015. Recruitment of juvenile, captive-reared eastern loggerhead shrikes Lanius ludovicianus migrans into the wild population in Canada. Oryx 49:321-328.

Laikre, L., S. Palm, and N. Ryman. 2005. Genetic population structure of fishes: implications for coastal. Ambio 34:111-119.

Lakoff, G. 2004. Don't Think of an Elephant! Chel-sea Green Publishing, White River Junction, VT.

Lambeck, R. 1997. Focal species: a multi-species umbrella for nature conservation. Conservation Biology 11:849-856.

Lamont, A. 2006. Policy characterization of ecosystem management. Environmental Monitoring and Assessment 113:5-18.

Lance, W.R., R. Clifton, and B. Ferguson. 2005. Recent Social Trends in Canada, 1960–2000. McGill-Queen's University Press, Montreal, QC.

Lande, R. 1993. Risks of population extinction from demographic and environmental stochasticity and random catastrophes. The American Naturalist 142:911-927.

Lande, R. and G. Barrowclough 1987. Effective population size, genetic variation, and their use in population management. Pages 87-124 in Viable Populations for Conservation, edited by M. Soule. Cambridge University Press, Cambridge, UK.

Lande, R. and S. Shannon. 1996. The role of genetic variation in adaptation and population persistence in a changing environment. Evolution 50:434-437.

Landhäusser, S., D. Deshaies, and V. Lieffers. 2010. Disturbance facilitates rapid range expansion of aspen into higher elevations of the Rocky Mountains under a warming climate. Journal of Biogeography 37:68-76.

Landres, P., P. Morgan, and F. Swanson. 1999. Overview of the use of natural variability concepts in managing ecological systems. Ecological Applications 9:1179-1188.

Lang, J. and W. Hallman. 2005. Who does the public trust? The case of genetically modified food in the United States. Risk Analysis 25:1241-1252.

Larkin, P. 1977. An epitaph for the concept of maximum sustained yield. Transactions of the American Fisheries Society 106:1-11.

Larsen, C. 1997. Spatial and temporal variations in boreal forest fire frequency in northern Alberta. Journal of Biogeography 24:663-673.

Latham, A., M. Latham, N. McCutchen, and S. Boutin. 2011. Invading white-tailed deer change wolf–caribou dynamics in northeastern Alberta. The Journal of Wildlife Management 75:204-212.

Laurance, W. 2008. Theory meets reality: how habitat fragmentation research has transcended island biogeographic theory. Biological Conservation 141:1731-1744.

Lawler, J. 2009. Climate change adaptation strategies for resource management and conservation planning. Annals of the New York Academy of Sciences 1162:79-98.

Lazar, A. 2003. The Canadian forest products industry's emerging industrial model for sustainability. The Forestry Chronicle 79:769-773.

Lee, A., B. Saether, and S. Engen. 2011. Demographic stochasticity, Allee effects, and extinction: the influence of mating system and sex ratio. The American Naturalist 177:301-313.

Lee, K. 1993. Compass and Gyroscope: Integrating Science and Politics for the Environment. Island Press, Washington, D.C.

Lee, P. and S. Boutin. 2006. Persistence and developmental transition of wide seismic lines in the western Boreal Plains of Canada. Journal of Environmental Management 78:240-250.

Lele, S., E. Merrill, J. Keim, and M. Boyce. 2013. Selection, use, choice and occupancy: clarifying concepts in resource selection studies. Journal of Animal Ecology 82:1183-1191.

Lemieux, C., T. Beechey, D. Scott, and P. Gray. 2010. Protected Areas and Climate Change in Canada Challenges and Opportunities for Adaptation. Occasional Paper No. 19. Canadian Council on Ecological Areas, Ottawa, ON.

Lemieux, C., T. Beechey, D. Scott, and P. Gray. 2011. The state of climate change adaptation in Canada's protected areas sector. The Canadian Geographer 55:301-317.

Lenoir, J., T. Hattab, and G. Pierre. 2017. Climatic microrefugia under anthropogenic climate change: implications for species redistribution. Ecography 40:253—266.

Leopold, A. 1949. A Sand County Almanac. Oxford University Press, Oxford, UK.

Leroux, S., F. Schmiegelow, R. Lessard, and S. Cumming. 2007. Minimum dynamic reserves: a framework for determining reserve size in ecosystems structured by large disturbances. Biological Conservation 138:464-473.

Levin, S. 1970. Community equilibria and stability, and an extension of the competitive exclusion principle. The American Naturalist 104:413-423.

Li, H., X. Wang, and A. Hamann. 2010. Genetic adaptation of aspen (Populus tremuloides) populations to spring risk environments: a novel remote sensing approach. Canadian Journal of Forest Research 40:2082-2090.

Lieffers, V. and J. Beck. 1994. A semi-natural approach to mixedwood management in the prairie provinces. The Forestry Chronicle 70:260-264.

Lindenmayer, D., J. Fischer, and R. Cunningham. 2005. Native vegetation cover thresholds associated with species responses. Biological Conservation 124:311-316.

Lindenmayer, D. and G. Likens. 2011. Direct measurement versus surrogate indicator species for evaluating environmental change and biodiversity loss. Ecosystems 14:47-59.

Lindenmayer, D., W. Steffen, A. Burbidge, L. Hughes, and R. Kitching. 2010. Conservation strategies in response to rapid climate change: Australia as a case study. Biological Conservation 143:1587-1593.

Littlefield, C., B. McRae, J. Michalak, J. Lawler, and C. Carroll. 2017. Connecting today's climates to future climate analogs to facilitate movement of species under climate change. Conservation Biology 31:1397-1408.

Loarie, S., P. Duffy, H. Hamilton, G. Asner, and C. Field. 2009. The velocity of climate change. Nature 462:1052-1055.

Locke, H. 2014. Nature needs half: a necessary and hopeful new agenda for protected areas in North America and around the world. The George Wright Forum 31:359-371.

Loehle, C. and D. Sleep. 2015. Use and application of range mapping in assessing extinction risk in Canada. Wildlife Society Bulletin 39:658-663.

Long, J. 2009. Emulating natural disturbance regimes as a basis for forest management: a North American view. Forest Ecology and Management 257:1868-1873.

Long, R., A. Charles, and R. Stephenson. 2015. Key principles of marine ecosystem-based management. Marine Policy 57:53-60.

Lonsdale, W., H. Kretser, C. Chetkiewicz, and M. Cross. 2017. Similarities and differences in barriers and opportunities affecting climate change adaptation action in four North American landscapes. Environmental Management 60:1076-1089.

Loo, T. 2006. States of Nature. UBC Press, Vancouver, BC.

Louda, S.,. Kendall,. Connor, and D. Simberloff. 1997. Ecological effects of an insect introduced for the biological control of weeds. Science 277:1088-1090.

Luckert, M., D. Haley, and G. Hoberg. 2011. Policies for Sustainably Managing Canada's Forests: Tenure, Stumpage Fees, and Forest Practices. UBC Press, Vancouver, BC.

Lüthi, D., M. Le Floch, B. Bereiter, T. Blunier, and J. Barnola. 2008. High-resolution carbon dioxide concentration record 650,000–800,000 years before present. Nature 453:379-382.

MacArthur, R. and E. Wilson. 1967. The Theory of Island Biogeography. Princeton University Press, Princeton, NJ.

Mac Nally, R., A. Bennett, G. Brown, L. Lumsden, and A. Yen. 2002. How well do ecosystem-based planning units represent different components of biodiversity? Ecological Applications 12:900-912.

MacCleery, D. 2008. Re-inventing the United States Forest Service: evolution of national forests from custodial management, to production forestry, to ecosystem management. Pages 45-78 in Re-inventing Forestry Agencies: Experiences of Institutional Restructuring in Asia and the Pacific, edited by P. Durst, C. Brown, J. Broadhead, R. Suzuki, and R. Leslie. Food And Agriculture Organization Of The United Nations, Bangkok, Thailand.

MacDowell, L.S. 2012. An Environmental History of Canada. UBC Press, Vancouver, BC.

Mace, G., N. Collar, K. Gaston, C. Hilton-Taylor, and H. Akakaya. 2008. Quantification of extinction risk: IUCN's system for classifying threatened species. Conservation Biology 22:1424-1442.

MacEachern, A. 2003. The conservation movement. Pages 1-16 in Canada: Confederation to Present, edited by B. Hesketh and C. Hackett. Chinook Multimedia, Edmonton, AB.

MacEwan, G. 1995. Buffalo—Sacred and Sacrificed. Alberta Sport, Recreation, Parks and Wildlife Foundation, Edmonton, AB.

MacKasey, W. 2000. Abandoned Mines in Canada. WOM Geological Associates Inc., Sudbury, ON.

MacKenzie, D. and J. Royle. 2005. Designing occupancy studies: general advice and allocating survey effort. Journal of Applied Ecology 42:1105-1114.

MacMillan, D. and K. Marshall. 2006. The Delphi process: an expert-based approach to ecological modelling in data-poor environments. Animal Conservation 9:11-19.

MacNearney, D., K. Pigeon, G. Stenhouse, W. Nijland, and N. Coops. 2016. Heading for the hills? Evaluating spatial distribution of woodland caribou in response to a growing anthropogenic disturbance footprint. Ecology and Evolution 6:6484-6509.

Magness, D., A. Lovecraft, and J. Morton. 2012. Factors influencing individual management preferences for facilitating adaptation to climate change within the National Wildlife Refuge System. Wildlife Society Bulletin 36:457-468.

Malcolm, J., A. Markham, R. Neilson, and M. Garaci. 2002. Estimated migration rates under scenarios of global climate change. Journal of Biogeography 29:835-849.

Mallory, M., C. Ogilvie, and H. Gilchrist. 2006. A review of the Northern Ecosystem Initiative in Arctic Canada: facilitating Arctic ecosystem research through traditional and novel approaches. Environmental Monitoring and Assessment 113:19-29.

Mandrak, N. and B. Cudmore. 2010. The fall of native fishes and the rise of non-native fishes in the Great Lakes Basin. Aquatic Ecosystem Health & Management 13:255-268.

Margules, C. and R. Pressey. 2000. Systematic conservation planning. Nature 405:243-253.

Margules, C., R. Pressey, and P. Williams. 2002. Representing biodiversity: data and procedures for identifying priority areas for conservation. Journal of Biosciences 27:309-326.

Marion, J. and S. Reid. 2007. Minimising visitor impacts to protected areas: the efficacy of low impact education programmes. Journal of Sustainable Tourism 15:5-27.

Marshall, I., P. Schut, and M. Ballard. 1999. National Ecological Framework for Canada: Attribute Data. Agriculture and Agri-Food Canada, Ottawa, ON.

Martens, J., M. Entz, and M. Wonneck. 2013. Ecological Farming Systems on the Canadian Prairies: A Path to Profitability, Sustainability and Resilience. Agriculture and Agri-Food Canada, Ottawa, ON.

Martin, J., M. Runge, J. Nichols, B. Lubow, and W. Kendall. 2009. Structured decision making as a conceptual framework to identify thresholds for conservation and management. Ecological Applications 19:1079-1090.

Martin, M., E. Shorohova, and N. Fenton. 2023. Embracing the complexity and the richness of boreal old-growth forests: a further step toward their ecosystem management. Pages 191-218 in Boreal Forests in the Face of Climate Change, edited by M. Girona, S. Gauthier, H. Morin, and Y. Bergeron. Advances in Global Change Research, Volume 74.

Martin, T., A. Camaclang, H. Possingham, L. Maguire, and I. Chades. 2017. Timing of protection of critical habitat matters. Conservation Letters 10:308-316.

Martin, T., L. Kehoe, C. Mantyka-Pringle, I. Chades, and S. Wilson. 2018. Prioritizing recovery funding to maximize conservation of endangered species. Conservation Letters 11:e12604.

Martinez-Meyer, E., A. Townsend Peterson, and W. Hargrove. 2004. Ecological niches as stable distributional constraints on mammal species, with implications for Pleistocene extinctions and climate change projections for biodiversity. Global Ecology and Biogeography 13:305-314.

Martins, E., S. Hinch, S. Cooke, and D. Patterson. 2012. Climate effects on growth, phenology, and survival of sockeye salmon (Oncorhynchus nerka): a synthesis of the current state of knowledge and future research directions. Reviews in Fish Biology and Fisheries 22:887-914.

Marvier, M. 2014. New conservation is true conservation. Conservation Biology 28:1-3.

Maxwell, S., R. Fuller, T. Brooks, and J. Watson. 2016. Biodiversity: the ravages of guns, nets and bulldozers. Nature 536:143-145.

Mazur, A. 1998. Global environmental change in the news 1987–90 vs 1992–6. International Sociology 13:457-472.

McCallum, M. 2015. Vertebrate biodiversity losses point to a sixth mass extinction. Biodiversity and Conservation 24:2497-2519.

McCarthy, M. 2014. Contending with uncertainty in conservation management decisions. Annals of the New York Academy of Sciences 1322:77-91.

McCarthy, M. and H. Possingham. 2012. On triage, public values and informed debate: trade-offs around extinction. Decision Point 65:4-5.

McCauley, D. 2006. Selling out on nature. Nature 443:27-28.

McClay, A, K. Fry, E Korpela, R. Lange, and L. Roy. 2004. Costs and Threats of Invasive Species to Alberta's Natural Resources. Alberta Sustainable Resource Development, Edmonton, AB.

McCormack, P. 1992. The political economy of bison management in Wood Buffalo National Park. Arctic 367-380.

McCune, J., W. Harrower, S. Avery-Gomm, J. Brogan, and A. Csergo. 2013. Threats to Canadian species at risk: an analysis of finalized recovery strategies. Biological Conservation 166:254-265.

McDevitt-Irwin, J., S. Fuller, C. Grant, and J. Baum. 2015. Missing the safety net: evidence for inconsistent and insufficient management of at-risk marine fishes in Canada. Canadian Journal of Fisheries and Aquatic Sciences 72:1596-1608.

McDonald-Madden, E., P. Baxter, and H. Possingham. 2008. Subpopulation triage: how to allocate conservation effort among populations. Conservation Biology 22:656-665.

McFarlane, B., T. Beckley, E. Huddart-Kennedy, S. Nadeau, and S. Wyatt. 2011. Public views on forest management: value orientation and forest dependency as indicators of diversity. Canadian Journal of Forest Research 41:740-749.

McFarlane, B. and P. Boxall. 2000a. Factors influencing forest values and attitudes of two stakeholder groups: The case of the Foothills Model Forest, Alberta, Canada. Society & Natural Resources 13:649-661.

McFarlane, B. and P. Boxall. 2000b. Forest Values and Attitudes of the Public, Environmentalists, Professional Foresters, and Members of Public Advisory Groups in Alberta. Report NOR-X-374. Canadian Forest Service, Edmonton, AB.

McFarlane, B. and P. Boxall. 2003. The role of social psychological and social structural variables in environmental activism: an example of the forest sector. Journal of Environmental Psychology 23:79-87.

McFarlane, B. and L. Hunt. 2006. Environmental activism in the forest sector social psychological, social-cultural, and contextual effects. Environment and Behavior 38:266-285.

McFarlane, B, D. Watson, and R. Stumpf-Allen. 2007. Public Perceptions of Conservation of Grizzly Bears in the Foothills Model Forest: A Survey of Local and Edmonton Residents. Report NOR-X-413. Canadian Forest Service, Edmonton, AB.

McInerney, F. and S. Wing. 2011. The Paleocene-Eocene Thermal Maximum: a perturbation of carbon cycle, climate, and biosphere with implications for the future. Annual Review of Earth and Planetary Sciences 39:489-516.

McKenney, D., J. Pedlar, K. Lawrence, K. Campbell, and M. Hutchinson. 2007. Potential impacts of climate change on the distribution of North American trees. BioScience 57:939-948.

McKinley, D., A. Miller-Rushing, H. Ballard, R. Bonney, and H. Brown. 2017. Citizen science can improve conservation science, natural resource management, and environmental protection. Biological Conservation 208:15-28.

McLachlan, S. and D. Bazely. 2003. Outcomes of long-term deciduous forest restoration in southwestern Ontario, Canada. Biological Conservation 113:159-169.

McLachlan, S. and A. Knispel. 2005. Assessment of long-term tallgrass prairie restoration in Manitoba, Canada. Biological Conservation 124:75-88.

McShane, T., P. Hirsch, T. Trung, A. Songorwa, and A. Kinzig. 2011. Hard choices: making trade-offs between biodiversity conservation and human well-being. Biological Conservation 144:966-972.

Meghani, Z. and J. Kuzma. 2011. The "revolving door" between regulatory agencies and industry: A problem that requires reconceptualizing objectivity. Journal of Agricultural and Environmental Ethics 24:575-599.

Meine, C., M. Soule, and R. Noss. 2006. "A mission-driven discipline": the growth of conservation biology. Conservation Biology 20:631-651.

Meir, E., S. Andelman, and H. Possingham. 2004. Does conservation planning matter in a dynamic and uncertain world? Ecology Letters 7:615-622.

Mennechez, G., N. Schtickzelle, and M. Baguette. 2003. Metapopulation dynamics of the bog fritillary butterfly: comparison of demographic parameters and dispersal between a continuous and a highly fragmented landscape. Landscape Ecology 18:279-291.

Merrick, A. 2006. Victoria's Secret goes green on paper for catalogs. Wall Street Journal, online news article: www.wsj.com/articles/SB116545222061942857.

Messier, C., R. Tittler, D. Kneeshaw, N. Gelinas, and A. Paquette. 2009. TRIAD zoning in Quebec: experiences and results after 5 years. The Forestry Chronicle 85:885-896.

Meyer, J., P. Frumhoff, S. Hamburg, and C. de la Rosa. 2010. Above the din but in the fray: environ- mental scientists as effective advocates. Frontiers in Ecology and the Environment 8:299-305.

Michalak, J., C. Carroll, S. Nielsen, and J. Lawler. 2018. Land facet data for North America. Databasin website: https://adaptwest.databasin.org/pages/adaptwest-landfacets.

Miller, B., M. Soule, and J. Terborgh. 2014. 'New conservation'or surrender to development? Animal Conservation 17:509-515.

Millar, C., N. Stephenson, and S. Stephens. 2007. Climate change and forests of the future: managing in the face of uncertainty. Ecological Applications 17:2145-2151.

Mills, T. 2000. Position advocacy by scientists risks science credibility. Northwest Science 74:165-168.

Mills, T. and R. Clark. 2001. Roles of research scientists in natural resource decision-making. Forest Ecology and Management 153:189-198.

Mineau, P., C. Downes, D. Kirk, E. Bayne, and M. Csizy. 2005. Patterns of bird species abundance in relation to granular insecticide use in the Canadian prairies. Ecoscience 12:267-278.

Mitchell, A., T. Wellicome, D. Brodie, and K. Cheng. 2011. Captive-reared burrowing owls show higher site-affinity, survival, and reproductive performance when reintroduced using a soft-release. Biological Conservation 144:1382-1391.

Moehrenschlager, A. and C. Moehrenschlager. 2001. Census of Swift Fox (Vulpes velox) in Canada and Northern Montana: 2000-2001. Alberta Species at Risk Report No. 24. Alberta Sustainable Development, Edmonton, AB.

Moehrenschlager, A. and C. Moehrenschlager. 2018. Population Survey of Reintroduced Swift Foxes (Vulpes velox) in Canada and Northern Montana 2014/2015. Centre for Conservation Research, Calgary Zoological Society, Calgary, AB.

Mogensen, S., J. Post, and M. Sullivan. 2013. Vulnerability to harvest by anglers differs across climate, productivity, and diversity clines. Canadian Journal of Fisheries and Aquatic Sciences 71:416-426.

Moilanen, A., V. Veach, L. Meller, A. Arponen, and H. Kujala. 2014. Zonation. Spatial Conservation Planning Methods and Software, Version 4. User Manual. C-BIG Conservation Biology Informatics Group, University of Helsinki, Finland.

Moore, K., B. Hopkin, S. Morgan, T. Beckley, and P. Angelstam. 2005. SmartWood Certification Assessment Report for Alberta-Pacific Forest Industries Inc. Forest Management Agreement Area. Al-Pac, online report: https://alpac.ca/application/files/1614/1876/0528/2005_Final_Report.pdf.

Moore, S., F. Cubbage, and C. Eicheldinger. 2012. Impacts of forest stewardship council (FSC) and sustainable forestry initiative (SFI) forest certification in north America. Journal of Forestry 110:79-88.

Morelli, T., C. Daly, S. Dobrowski, D. Dulen, and J. Ebersole. 2016. Managing climate change refugia for climate adaptation. PloS One 11:e0159909.

Moritz, C. and R. Agudo. 2013. The future of species under climate change: resilience or decline? Science 341:504-508.

Morrison, C., C. Wardle, and J. Castley. 2016. Repeatability and reproducibility of population viability analysis (PVA) and the implications for threatened species management. Frontiers in Ecology and Evolution 4:98.

MSC. 2017. Global Impacts Report 2017. Marine Stewardship Council, online report: www.msc.org/.

Mueller, J. and J. Hellmann. 2008. An assessment of invasion risk from assisted migration. Conservation Biology 22:562-567.

Murcia, C., J. Aronson, G. Kattan, D. Moreno-Mateos, and K. Dixon. 2014. A critique of the 'novel ecosystem'concept. Trends in Ecology & Evolution 29:548-553.

Murdoch, W., S. Polasky, K. Wilson, H. Possingham, and P. Kareiva. 2007. Maximizing return on investment in conservation. Biological Conservation 139:375-388.

Mwale, D., T. Gan, K. Devito, C. Mendoza, and U. Silins. 2009. Precipitation variability and its relationship to hydrologic variability in Alberta. Hydrological Processes 23:3040-3056.

Myers, N., R. Mittermeier, C. Mittermeier, G. Da Fonseca, and J. Kent. 2000. Biodiversity hotspots for conservation priorities. Nature 403:853-858.

NABCI (North American Bird Conservation Initiative). 2012. The State of Canada's Birds: 2012. Environment Canada, Ottawa, ON.

Nadasty, P. 2003. Reevaluating the co-management success story. Arctic 56:367-380.

Naidoo, R., A. Balmford, P. Ferraro, S. Polasky, and T. Ricketts. 2006. Integrating economic costs into conservation planning. Trends in Ecology & Evolution 21:681-687.

NASA. 2023. Global Climate Change: Vital Signs of the Planet. NASA, online data retrieved Feb. 22, 2023 from https://climate.nasa.gov/vital-signs

Natcher, D., S. Davis, and C. Hickey. 2005. Co-management: managing relationships, not resources. Human Organization 64:240-250.

NAWMPC. 2012. North American Waterfowl Management Plan 2012: People Conserving Waterfowl and Wetlands. North American Waterfowl Management Plan Committee, online report: http://nawmp.wetlandnetwork.ca

NCC. 2017. 2016–2017 Conservation Impact Through Partnership: The Natural Areas Conservation Program. Nature Conservancy of Canada, online report: www.natureconservancy.ca.

Neil, D. 2015. VW lost its moral compass in quest for growth. Wall Street Journal, online news article: www.wsj.com/articles/vw-lost-its-moral-compass-in-quest-for-growth-1443115862.

Nelson, M. and J. Vucetich. 2009. On advocacy by environmental scientists: what, whether, why, and how. Conservation Biology 23:1090-1101.

Nesbitt, L. and J. Adamczewski. 2013. Decline and Recovery of the Bathurst Caribou Herd: Workshops Held in Yellowknife, NWT, October 1 and 2, and 5 and 6, 2009. Government of NWT online report: www.enr.gov.nt.ca/sites/enr/files/manuscript_reports/238_manuscript.pdf.

Nielsen, J., G. Ruggerone, and C. Zimmerman. 2013. Adaptive strategies and life history characteristics in a warming climate: Salmon in the Arctic? Environmental Biology of Fishes 96:1187-1226.

Nielsen, S., G. Stenhouse, and M. Boyce. 2006. A habitat-based framework for grizzly bear conservation in Alberta. Biological Conservation 130:217-229.

Nitschke, C. 2008. The cumulative effects of resource development on biodiversity and ecological integrity in the Peace-Moberly region of Northeast British Columbia, Canada. Biodiversity and Conservation 17:1715-1740.

Nitschke, C. and J. Innes 2004. The application of forest zoning as an alternative to multiple use forestry. Pages 97-124 in Forestry and Environmental Change: Socioeconomic and Political Dimensions: Report No. 5 of the IUFRO Task Force on Environmental Change, edited by J. Innes, G. Hickey, and H. Hoen. CABI Publishing, New York, NY.

Nixon, A., R. J. Fisher, D. Stralberg, E. Bayne, and D. Farr. 2016. Projected responses of North American grassland songbirds to climate change and habitat availability at their northern range limits in Alberta, Canada. Avian Conservation and Ecology 11:(2):2.

Noble, B., J. Liu, and P. Hackett. 2017. The contribution of project environmental assessment to assessing and managing cumulative effects: individually and collectively insignificant? Environmental Management 59:531-545.

Noble, B., P. Sheelanere, and R. Patrick. 2011. Advancing watershed cumulative effects assessment and management: lessons from the South Saskatchewan River watershed, Canada. Journal of Environmental Assessment Policy and Management 13:567-590.

Noga, W. and W. Adamowicz. 2014. A Study of Canadian Conservation Offset Programs: Lessons Learned from a Review of Programs, Analysis of Stakeholder Perceptions, and Investigation of Transactions Costs. Sustainable Prosperity, University of Ottawa, Ottawa, ON.

Northrup, J., J. Pitt, T. Muhly, G. Stenhouse, and M. Musiani. 2012. Vehicle traffic shapes grizzly bear behaviour on a multiple-use landscape. Journal of Applied Ecology 49:1159-1167.

Noss, R. 2000. High-risk ecosystems as foci for considering biodiversity and ecological integrity in ecological risk assessments. Environmental Science & Policy 3:321-332.

Noss, R. 2001. Beyond Kyoto: forest management in a time of rapid climate change. Conservation Biology 15:578-590.

Noss, R. 2003. A checklist for wildlands network designs. Conservation Biology 17:1270-1275.

Noss, R. 2007. Values are a good thing in conservation biology. Conservation Biology 21:18-20.

Noss, R., A. Dobson, R. Baldwin, P. Beier, and C. Davis. 2012. Bolder thinking for conservation. Conservation Biology 26:1-4.

NRC (Natural Regions Committee). 2006. Natural Regions and Subregions of Alberta. Government of Alberta, Edmonton, AB.

NRCAN. 2013. Additional Statistics—Minerals and Metals. Natural Resources Canada, online report: www.nrcan.gc.ca/publications/statistics-facts/1245.

NRCAN. 2018. 10 Key Facts on Canada's Natural Resources. Natural Resources Canada, online report: www.nrcan.gc.ca/publications/key-facts/16013.

Nriagu, J., H. Wong, G. Lawson, and P. Daniel. 1998. Saturation of ecosystems with toxic metals in Sudbury basin, Ontario, Canada. Science of the Total Environment 223:99-117.

NRTEE. 2003. Securing Canada's Natural Capital: a Vision for Nature Conservation in the 21st Century. National Round Table on the Environment and the Economy, Ottawa, ON.

Nunes, P. and J. van den Bergh. 2001. Economic valuation of biodiversity: sense or nonsense? Ecological Economics 39:203-222.

Nunez, T., J. Lawler, B. McRae, D. Pierce, and M. Krosby. 2013. Connectivity planning to address climate change. Conservation Biology 27:407-416.

NWSAR. 2016. Our Petition—Making our Voices Heard. Northwest Species at Risk Committee, Online press release: https://albertanwsar.ca.

Nyland, R.D. 1996. Silviculture Concepts and Applications. McGraw-Hill Co., New York, NY.

O'Neill, G., T. Wang, N. Ukrainetz, L. Charleson, and L. McAuley. 2017. A Proposed Climate-Based Seed Transfer System for British Columbia. Government of BC, Victoria, BC.

O'Rourke, D. 2005. Market movements—nongovernmental organization strategies to influence global production and consumption. Journal of Industrial Ecology 9:115-128.

OBC. 2011. Ontario's Biodiversity Strategy, 2011: Renewing Our Commitment to Protecting What Sustains Us. Ontario Biodiversity Council, Peterborough, ON.

Olive, A. 2015. Urban and rural attitudes toward endangered species conservation in the Canadian prairies: drawing lessons from the American ESA. Human Dimensions of Wildlife 20:189-205.

Olive, A. 2016. It is just not fair: the Endangered Species Act in the United States and Ontario. Ecology and Society 21:13.

Olive, A. and J. McCune. 2017. Wonder, ignorance, and resistance: landowners and the stewardship of endangered species. Journal of Rural Studies 49:13-22.

Oliver, T., T. Brereton, and D. Roy. 2013. Population resilience to an extreme drought is influenced by habitat area and fragmentation in the local landscape. Ecography 36:579-586.

OMNR. 2009. Forest Management Planning Manual for Ontario's Crown Forests. Ontario Ministry of Natural Resources, Toronto, ON.

OMNR. 2012. Biodiversity: It's in Our Nature. Ontario Government Plan to Conserve Biodiversity 2012-2020. Ontario Ministry of Natural Resources, Toronto, ON.

OMNRF. 2015. Far North Land Use Strategy: A Draft. Ontario Ministry of Natural Resources and Forestry, Toronto, ON.

Osko, T. and M. Glasgow. 2010. Removing the Wellsite Footprint: Recommended Practices for Construction and Reclamation of Wellsites on Upland Forests in Boreal Alberta. University of Alberta online report: www.biology.ualberta.ca/faculty/stan_boutin/ilm/?Page=8162.

Ostrom, E., M. Janssen, and J. Anderies. 2007. Going beyond panaceas. Proceedings of the National Academy of Sciences 104:15176-15178.

OWA. 2022. Annual Report 2021/22. Orphan Well Association, Calgary, AB.

Pacala, S. and R. Socolow. 2004. Stabilization wedges: solving the climate problem for the next 50 years with current technologies. Science 305:968-972.

Paris, M. 2012. Caribou Recovery Plan Swamped by Public Feedback. CBC, online news article: www.cbc.ca/news.

Park, D., M. Sullivan, E. Bayne, and G. Scrimgeour. 2008. Landscape-level stream fragmentation caused by hanging culverts along roads in Alberta's boreal forest. Canadian Journal of Forest Research 38:566-575.

Parkins, J, R Stedman, and B McFarlane. 2001. Public Involvement in Forest Management and Planning: a Comparative Analysis of Attitudes and Preferences in Alberta. Report NOR-X-382. Canadian Forest Service, Edmonton, AB.

Parlee, B., J. Sandlos, and D. Natcher. 2018. Undermining subsistence: barren-ground caribou in a “tragedy of open access”. Science Advances 4:e1701611.

Parmesan, C., T. Root, and M. Willig. 2000. Impacts of extreme weather and climate on terrestrial biota. Bulletin of the American Meteorological Society 81:443-450.

Parr, M., L. Bennun, T. Boucher, T. Brooks, and C. Chutas. 2009. Why we should aim for zero extinction. Trends in Ecology & Evolution 24:181.

Parsons, M. 2007. Reducing Risks From Metals: Environmental Impacts of Historical Gold Mines. Natural Resources Canada, Ottawa, ON.

Passelac-Ross, M. 2006. Overview of Provincial Wildlife Laws. Canadian Institute of Resources Law, Calgary, AB.

Patry, C., D. Kneeshaw, S. Wyatt, F. Grenon, and C. Messier. 2013. Forest ecosystem management in North America: from theory to practice. The Forestry Chronicle 89:525-537.

Pauly, D. 1995. Anecdotes and the shifting baseline syndrome of fisheries. Trends in Ecology & Evolution 10:430.

PC. 1997. National Park System Plan. Parks Canada, Ottawa, ON.

PC. 2008. Principles and Guidelines for Ecological Restoration in Canada’s Protected Natural Areas. Parks Canada, Ottawa, Quebec.

PC. 2016. Multi-Species Action Plan for Gros Morne National Park of Canada. Parks Canada, Ottawa, ON.

Pearson, A. 2010. Natural and logging disturbances in the temperate rain forests of the Central Coast, British Columbia. Canadian Journal of Forest Research 40:1970-1984.

Pearson, R. 2006. Climate change and the migration capacity of species. Trends in Ecology & Evolution 21:111-113.

Pearson, R. and T. Dawson. 2003. Predicting the impacts of climate change on the distribution of species: are bioclimate envelope models useful? Global Ecology & Biogeography 12:361-371.

Pearson, R., J. Stanton, K. Shoemaker, M. Aiello-Lammens, and P. Ersts. 2014. Life history and spatial traits predict extinction risk due to climate change. Nature Climate Change 4:217-221.

Pelletier, D., M. Clark, M. Anderson, B. Rayfield, and M. Wulder. 2014. Applying circuit theory for corridor expansion and management at regional scales: tiling, pinch points, and omnidirectional connectivity. PLoS One 9:e84135.

Perera, A., B. Dalziel, L. Buse, and R. Routledge. 2009. Spatial variability of stand-scale residuals in Ontario’s boreal forest fires. Canadian Journal of Forest Research 39:945-961.

Perez, I., J. Anadon, M. Diaz, G. Nicola, and J. Tella. 2012. What is wrong with current translocations? A review and a decision-making proposal. Frontiers in Ecology and the Environment 10:494-501.

Perkl, R., R. Baldwin, S. Trombulak, and G. Smith. 2016. Landscape network congruency at the ecoregion scale: can we expect corridor ‘umbrellas’? Journal of Land Use Science 11:429-449.

Peters, W., M. Hebblewhite, N. DeCesare, F. Cagnacci, and M. Musiani. 2013. Resource separation analysis with moose indicates threats to caribou in human altered landscapes. Ecography 36:487-498.

Peterson, G., G. Cumming, and S. Carpenter. 2003. Scenario planning: a tool for conservation in an uncertain world. Conservation Biology 17:358-366.

Phillips, S., M. McCuaig-Boyd, and O. Carlier. 2018. Sessional Paper 17-2018; 4th Session, 29th Legislature. Government of Alberta, Edmonton, AB.

Pickell, P., D. Andison, N. Coops, S. Gergel, and P. Marshall. 2015. The spatial patterns of anthropogenic disturbance in the western Canadian boreal forest following oil and gas development. Canadian Journal of Forest Research 45:732-743.

Pickering, C. 2010. Ten factors that affect the severity of environmental impacts of visitors in protected areas. Ambio 39:70-77.

Pielou, E.C. 1991. After the Ice Age: The Return of Life to Glaciated North America. University of Chicago Press, Chicago, IL.

PM (Panterra Management). 2009. Exploring Integrated Landscape Management at the Provincial Level in Canada. Government of Canada Policy Research Initiative, Environment Canada, Ottawa, ON

Poiani, K., R. Goldman, J. Hobson, J. Hoekstra, and K. Nelson. 2011. Redesigning biodiversity conservation projects for climate change: examples from the field. Biodiversity and Conservation 20:185-201.

Posewitz, J. 1994. Beyond Fair Chase: The Ethic and Tradition of Hunting. Globe Pequot Press, Guilford, Connecticut.

Possingham, H., S. Andelman, M. Burgman, R. Medellin, and L. Master. 2002. Limits to the use of threatened species lists. Trends in Ecology & Evolution 17:503-507.

Post, J., N. Mandrak, and M. Burridge 2016. Canadian freshwater fishes, fisheries and their management, south of 60 N. Pages 151-165 in Freshwater Fisheries Ecology. Wiley Blackwell, West Sussex, UK.

Post, J., M. Sullivan, S. Cox, N. Lester, and C. Walters. 2002. Canada's recreational fisheries: the invisible collapse? Fisheries 27:6-17.

Poulin, J., M. Villard, M. Edman, P. Goulet, and A. Eriksson. 2008. Thresholds in nesting habitat requirements of an old forest specialist, the Brown Creeper (Certhia americana), as conservation targets. Biological Conservation 141:1129-1137.

Pratt, L. and I. Urquhart. 1994. The Last Great Forest: Japanese Multinationals and Alberta's Northern Forests. NeWest Press, Edmonton, AB.

Pressey, R., M. Watts, M. Ridges, and T. Barrett. 2005. C-Plan Conservation Planning. Department of Environment and Conservation, Government of Western Australia.

Price, D., R. Alfaro, K. Brown, M. Flannigan, and R. Fleming. 2013. Anticipating the consequences of climate change for Canada's boreal forest ecosystems. Environmental Reviews 21:322-365.

Probert, W., C. Hauser, E. McDonald-Madden, M. Runge, and P. Baxter. 2011. Managing and learning with multiple models: objectives and optimization algorithms. Biological Conservation 144:1237-1245.

Proctor, M., S. Nielsen, W. Kasworm, C. Servheen, and T. Radandt. 2015. Grizzly bear connectivity mapping in the Canada–United States trans-border region. The Journal of Wildlife Management 79:544-558.

Proulx, G., S. Alexander, H. Barron, M. Bekoff, and R. Brook. 2017. Killing wolves and farming caribou benefit industry, not caribou: a response to Stan Boutin. Nature Alberta 47:4-11.

Pruss, S., P. Fargey, and A. Moehrenschlager. 2008. Recovery Strategy for the Swift Fox (Vulpes velox) in Canada. Species at Risk Act Recovery Strategy Series. Parks Canada, Ottawa, ON.

Pulliam, H. and B. Danielson. 1991. Sources, sinks, and habitat selection: a landscape perspective on population dynamics. The American Naturalist 137:S50-S66.

Pullin, A., T. Knight, D. Stone, and K. Charman. 2004. Do conservation managers use scientific evidence to support their decision-making? Biological Conservation 119:245-252.

PWPC. 2011. Final Recommended Peel Watershed Regional Land Use Plan. Peel Watershed Planning Commission, Whitehorse, YT.

Pybus, M. and P. Rowell 2005. Fisheries: research and management. Pages 177-216 in Fish, Fur and Feathers: Fish and Wildlife Conservation in Alberta 1905–2005. Fish and Wildlife Historical Society, Edmonton, AB.

Raftery, A., A. Zimmer, D. Frierson, R. Startz, and P. Liu. 2017. Less than 2 C warming by 2100 unlikely. Nature Climate Change 7:637-641.

Raiter, K., S. Prober, H. Possingham, F. Westcott, and R. Hobbs. 2018. Linear infrastructure impacts on landscape hydrology. Journal of Environmental Management 206:446-457.

Rayfield, B., P. James, A. Fall, and M. Fortin. 2008. Comparing static versus dynamic protected areas in the Quebec boreal forest. Biological Conservation 141:438-449.

Rayner, J. and M. Howlett. 2009. Conclusion: governance arrangements and policy capacity for policy integration. Policy and Society 28:165-172.

RBC. 2015. Socially Responsible Investing. Royal Bank of Canada, online report: http://funds.rbcgam.com/investment-solutions/socially-responsible-investments/index.html.

Redford, K. and W. Adams. 2009. Payment for ecosystem services and the challenge of saving nature. Conservation Biology 23:785-787.

Reed, M., L. Stringer, I. Fazey, A. Evely, and J. Kruijsen. 2014. Five principles for the practice of knowledge exchange in environmental management. Journal of Environmental Management 146:337-345.

Reed, T., D. Schindler, and R. Waples. 2011. Interacting effects of phenotypic plasticity and evolution on population persistence in a changing climate. Conservation Biology 25:56-63.

Reese, S. 2001. Prologue – framing public life. Pages 7-31 in Framing Public Life: Perspectives on Media and Our

Understanding of the Social World, edited by S. Reese, O. Gandy, and A. Grant. Lawrence Erlbaum Associates Inc., Mahwah, NJ.

Rehfeldt, G., N. Crookston, C. Saenz-Romero, and E. Campbell. 2012. North American vegetation model for land-use planning in a changing climate: a solution to large classification problems. Ecological Applications 22:119-141.

Reid, W., A. Mooney, D. Cropper, S. Carpenter, K. Chopra, and P. Dasgupta. 2005. Ecosystems and Human Well-Being: Synthesis. A Report of the Millenium Ecosystem Assessment. Island Press, Washington, DC.

Reining, C., K. Beazley, P. Doran, and C. Bettigole. 2006. From the Adirondacks to Acadia: A Wildlands Network Design for the Greater Northern Appalachians. Special Paper No. 7. Wildlands Project, Richmond, VT.

Rejmanek, M. and M. Pitcairn 2002. When is eradication of exotic pest plants a realistic goal? Pages 249-253 in Turning the Tide: the Eradication of Invasive Species, edited by C. Veitch and M. Clout. IUCN SSC Invasive Species Specialist Group, Gland Switzerland.

Ricciardi, A. and D. Simberloff. 2009. Assisted colonization is not a viable conservation strategy. Trends in Ecology & Evolution 24:248-253.

Richmond, S., E. Jenkins, A. Couturier, and M. Cadman. 2015. Thresholds in forest bird richness in response to three types of forest cover in Ontario, Canada. Landscape Ecology 30:1273-1290.

Rieger, S. 2018. Alberta backs down on plans to ban trout fishing from some rivers. CBC, online news article: www.cbc.ca/news/canada/calgary/alberta-trout-fishing-recovery-plan-1.4546810.

Riley, J., S. Green, and K. Brodribb. 2007. A Conservation Blueprint for Canada's Prairies and Parklands. The Nature Conservancy of Canada, Toronto, ON.

Robillard, C., L. Coristine, R. Soares, and J. Kerr. 2015. Facilitating climate-change-induced range shifts across continental land-use barriers. Conservation Biology 29:1586-1595.

Robinson, C., P. Duinker, and K. Beazley. 2010. A conceptual framework for understanding, assessing, and mitigating ecological effects of forest roads. Environmental Reviews 18:61-86.

Robinson, J. 2006. Conservation biology and real-world conservation. Conservation Biology 20:658-669.

Robson, M., A. Hawley, and D. Robinson. 2000. Comparing the social values of forest-dependent, provincial and national publics for socially sustainable forest management. The Forestry Chronicle 76:615-622.

Rocchini, D., G. Foody, H. Nagendra, C. Ricotta, and M. Anand. 2013. Uncertainty in ecosystem mapping by remote sensing. Computers & Geosciences 50:128-135.

Rochman, C., M. Browne, A. Underwood, J. Van Franeker, and R. Thompson. 2016. The ecological impacts of marine debris: unraveling the demonstrated evidence from what is perceived. Ecology 97:302-312.

Rönnqvist, M. 2003. Optimization in forestry. Mathematical Programming 97:267-284.

Rosatte, R. 2014. The behaviour and dynamics of a restored elk (Cervus canadensis manitobensis) population in southern Ontario, Canada: 5–12 years post restoration. Canadian Wildlife Biology & Management 3:34-51.

Rose, N. and P. Burton. 2009. Using bioclimatic envelopes to identify temporal corridors in support of conservation planning in a changing climate. Forest Ecology and Management 258:S64-S74.

Rose, M. and L. Hermanutz. 2004. Are boreal ecosystems susceptible to alien plant invasion? Evidence from protected areas. Oecologia 139:467-477.

Ross, M. 1997. A History of Forest Legislation in Canada 1867–1996. Canadian Institute of Resources Law, Calgary, AB.

Rothley, K. 2006. Finding the tradeoffs between the reserve design and representation. Environmental Management 38:327-337.

Rouget, M., R. Cowling, A. Lombard, A. Knight, and G. Kerley. 2006. Designing large-scale conservation corridors for pattern and process. Conservation Biology 20:549-561.

Rouget, M., R. Cowling, R. Pressey, and D. Richardson. 2003. Identifying spatial components of ecological and evolutionary processes for regional conservation planning in the Cape Floristic Region, South Africa. Diversity and Distributions 9:191-210.

Rudd, M., S. Andres, and M. Kilfoil. 2016. Non-use economic values for little-known aquatic species at risk: comparing choice experiment results from surveys focused on species, guilds, and ecosystems. Environmental Management 58:476-490.

Rudnick, D., S. Ryan, P. Beier, S. Cushman, and F. Dieffenbach. 2012. The Role of Landscape Connectivity in Planning and Implementing Conservation and Restoration Priorities. Issues in Ecology. Report No. 16. Ecological Society of America. Washington, DC.

Ruiz-Gutierrez, V., E. Bjerre, M., Otto, G. Zimmerman, and B. Millsap. 2021. A pathway for citizen science data to inform policy: A case study using eBird data for defining low-risk collision areas for wind energy development. Journal of Applied Ecology 58:1104-1111.

Runge, M. 2011. An introduction to adaptive management for threatened and endangered species. Journal of Fish and Wildlife Management 2:220-233.

Ryder, O.A. 1986. Species conservation and systematics: the dilemma of subspecies. Trends in Ecology & Evolution 1:9-10.

Saccheri, I., M. Kuussaari, M. Kankare, P. Vikman, and I. Hanski. 1998. Inbreeding and extinction in a butterfly metapopulation. Nature 392:491.

Sachsman, D. 1996. The mass media “discover” the environment: influences on environmental reporting in the first twenty years. Pages 241-256 in The Symbolic Earth: Discourse and Our Creation of the Environment, edited by J. Cantrill and C. Oravec. University Press of Kentucky, Lexington, KY.

Sandler, R. 2013. Climate change and ecosystem management. Ethics, Policy & Environment 16:1-15.

Sandlos, J. 2013. Nature’s nations: the shared conservation history of Canada and the USA. International Journal of Environmental Studies 70:358-371.

Sarkar, S., J. Justus, T. Fuller, C. Kelley, and J. Garson. 2005. Effectiveness of environmental surrogates for the selection of conservation area networks. Conservation Biology 19:815-825.

Sarkar, S., R. Pressey, D. Faith, C. Margules, and T. Fuller. 2006. Biodiversity conservation planning tools: present status and challenges for the future. Annual Review of Environmental Resources 31:123-159.

Sauer, J. and W. Link. 2011. Analysis of the North American breeding bird survey using hierarchical models. The Auk 128:87-98.

Savage, J. and M. Vellend. 2015. Elevational shifts, biotic homogenization and time lags in vegetation change during 40 years of climate warming. Ecography 38:546-555.

Savolainen, O., T. Pyhajarvi, and T. Knurr. 2007. Gene flow and local adaptation in trees. Annual Review of Ecology, Evolution, and Systematics 38:595-619.

SC. 1983a. Historical Statistics of Canada, Section M: Agriculture (11-516-X). Statistics Canada, online dataset: www.statcan.gc.ca/pub/11-516-x/3000140-eng.htm.

SC. 1983b. Historical Statistics of Canada, Section L: Lands (11-516-X). Statistics Canada, online dataset: www.statcan.gc.ca/pub/11-516-x/3000140-eng.htm.

SC. 2011. Human Activity and the Environment: Detailed Statistics 2011. Catalogue no. 16-201-S. Statistics Canada, Ottawa, ON.

SC. 2013. Human Activity and the Environment: Measuring Ecosystem Goods and Services in Canada. Statistics Canada, Ottawa, ON.

SC. 2014a. Estimated Population of Canada, 1605 to Present. Statistics Canada, online dataset: www.statcan.gc.ca/pub/98-187-x/4151287-eng.htm.

SC. 2014b. Human Activity and the Environment: Agriculture in Canada. Statistics Canada, Ottawa, ON.

SC. 2015. Population and Dwelling Count Highlight Tables, 2011 Census. Statistics Canada, online dataset: www.statcan.gc.ca/eng/start

SCC. 2017a. Chippewas of the Thames First Nation v. Enbridge Pipelines Inc., SCC 41. Supreme Court of Canada, Ottawa, ON.

SCC. 2017b. First Nation of Nacho Nyak Dun et al. v. Government of Yukon, SCC 58. Supreme Court of Canada, Ottawa, ON.

Schamberger, M., A. Farmer, and J. Terrell. 1982. Habitat Suitability Models: Introduction. US Fish and Wildlife Service, Fort Collins, CO.

Scheller, R., J. Domingo, B. Sturtevant, J. Williams, and A. Rudy. 2007. Design, development, and application of LANDIS-II, a spatial landscape simulation model with flexible temporal and spatial resolution. Ecological Modelling 201:409-419.

Schepers, D. 2010. Challenges to legitimacy at the Forest Stewardship Council. Journal of Business Ethics 92:279-290.

Schindler, D. 2001. The cumulative effects of climate warming and other human stresses on Canadian freshwaters in the new millennium. Canadian Journal of Fisheries and Aquatic Sciences 58:18-29.

Schlaepfer, M., W. Helenbrook, K. Searing, and K. Shoemaker. 2009. Assisted colonization: evaluating contrasting management actions (and values) in the face of uncertainty. Trends in Ecology & Evolution 24:471-472.

Schmidt, C., A. Mussell, J. Sweetland, and B. Seguin. 2012. The Greening of Canadian Agriculture: Policies to Assist Farmers as Stewards of the Environment. MacDonald-Laurier Institute, Ottawa, ON.

Schmiegelow, F. and M. Monkkonen. 2002. Habitat loss and fragmentation in dynamic landscapes: avian perspectives from the boreal forest. Ecological Applications 12:375-389.

Schneider, R. 2002. Alternative Futures: Alberta's Boreal Forest at the Crossroads. Federation of Alberta Naturalists, Edmonton, AB.

Schneider, R. 2013. Alberta's Natural Subregions Under a Changing Climate: Past, Present, and Future. Alberta Biodiversity Monitoring Institute, Edmonton, AB.

Schneider, R. and E. Bayne. 2015. Reserve design under climate change: from land facets back to ecosystem representation. PloS One 10:e0126918.

Schneider, R., K. Devito, N. Kettridge, and E. Bayne. 2016. Moving beyond bioclimatic envelope models: integrating upland forest and peatland processes to predict ecosystem transitions under climate change in the western Canadian boreal plain. Ecohydrology 9:899-908.

Schneider, R., G. Hauer, W.L. Adamowicz, and S. Boutin. 2010. Triage for conserving populations of threatened species: the case of woodland caribou in Alberta. Biological Conservation 143:1603-1611.

Schneider, R., G. Hauer, K. Dawe, W. Adamowicz, and S. Boutin. 2012. Selection of reserves for woodland caribou using an optimization approach. PloS one 7:e31672.

Schneider, R., G. Hauer, D. Farr, W. Adamowicz, and S. Boutin. 2011. Achieving conservation when opportunity costs are high: optimizing reserve design in Alberta's oil sands region. PloS one 6:e23254.

Schneider, R. and D. Pendlebury. 2016. Conservation Planning in Northwest Alberta. The Northern Alberta Conservation Area Working Group, online report: https://era.library.ualberta.ca/items/ec076fde-e280-445c-97e9-551696fccf49.

Schneider, R., J. Stelfox, S. Boutin, and S. Wasel. 2003. Managing the cumulative impacts of land uses in the Western Canadian Sedimentary Basin: a modeling approach. Conservation Ecology 7:8.

Schneider, R. and P. Yodzis. 1994. Extinction dynamics in the American marten (Martes americana). Conservation Biology 8:1058-1068.

Schreiber, S., C. Ding, A. Hamann, U. Hacke, and B. Thomas. 2013. Frost hardiness vs. growth performance in trembling aspen: an experimental test of assisted migration. Journal of Applied Ecology 50:939-949.

Schuetz, J., G. Langham, C. Soykan, C. Wilsey, and T. Auer. 2015. Making spatial prioritizations robust to climate change uncertainties: a case study with North American birds. Ecological Applications 25:1819-1831.

Schultz, J., E. Darling, and I. Cote. 2013. What is an endangered species worth? Threshold costs for protecting imperilled fishes in Canada. Marine Policy 42:125-132.

Schwartz, M., C. Brigham, J. Hoeksema, K. Lyons, and M. Mills. 2000. Linking biodiversity to ecosystem function: implications for conservation ecology. Oecologia 122:297-305.

Schwartz, M., J. Hiers, F. Davis, G. Garfin, and S. Jackson. 2017. Developing a translational ecology workforce. Frontiers in Ecology and the Environment 15:587-596.

Seidl, R. 2014. The shape of ecosystem management to come: anticipating risks and fostering resilience. BioScience 64:1159-1169.

Semeniuk, I. 2018. Too expensive to save? Why the best way to protect endangered species could mean letting some go. Globe and Mail, online news article: www.theglobeandmail.com/canada/article-too-expensive-to-save-new-approach-to-protecting-endangered-species/.

SERI. 2006. The SER International Primer on Ecological Restoration. Society for Ecological Restoration International, Tucson, AZ.

Sexton, J., P. McIntyre, A. Angert, and K. Rice. 2009. Evolution and ecology of species range limits. Annual Review of Ecology, Evolution, and Systematics 40:415-436.

Shaffer, M. 1981. Minimum population sizes for species conservation. BioScience 31:131-134.

Shang, J., J. Levesque, P. Howarth, B. Morris, and K. Staenz. 1999. Preliminary investigation of acid mine drainage detection using casi data, Copper Cliff, Ontario, Canada. The Fourth International Airborne Remote Sensing Conference and Exhibition, Ottawa, ON.

Shorohova, E., T. Aakala, S. Gauthier, D. Kneeshaw, and M. Koivula. 2023. Natural disturbances from the perspective of forest ecosystem-based management. Pages 88-121 in Boreal Forests in the Face of Climate Change, edited by M. Girona, S. Gauthier, H. Morin, and Y. Bergeron. Advances in Global Change Research, Volume 74.

SIF. 2022. 2022 Report on US Sustainable Investing Trends. Sustainable Investment Forum, online report: www.ussif.org//Files/Trends/2022/Trends%202022%20Executive%20Summary.pdf

Simberloff, D. 2015. Non-native invasive species and novel ecosystems. F1000Prime Reports 7:47.

Simberloff, D. and L. Abele. 1982. Refuge design and island biogeographic theory: effects of fragmentation. The American Naturalist 120:41-50.

Simberloff, D., J. Martin, P. Genovesi, V. Maris, and D. Wardle. 2013. Impacts of biological invasions: what's what and the way forward. Trends in Ecology & Evolution 28:58-66.

Simberloff, D. and B. Von Holle. 1999. Positive interactions of nonindigenous species: invasional meltdown? Biological Invasions 1:21-32.

Sissenwine, M. 1978. Is MSY an adequate foundation for optimum yield? Fisheries 3:22-42.

Small, E. 2011. The new Noah's Ark: beautiful and useful species only. Part 1. Biodiversity conservation issues and priorities. Biodiversity 12:232-247.

Smallwood, K. 2003. A Guide to Canada's Species at Risk Act. Sierra Legal Defence Fund, Vancouver BC.

Smeeton, C. and K. Weagle. 2000. The reintroduction of the swift fox Vulpes velox to South Central Saskatchewan, Canada. Oryx 34:171-179.

Smith, A., N. Hewitt, N. Klenk, D. Bazely, and N. Yan. 2012. Effects of climate change on the distribution of invasive alien species in Canada: a knowledge synthesis of range change projections in a warming world. Environmental Reviews 20:1-16.

Smith, A., N. Koper, C. Francis, and L. Fahrig. 2009. Confronting collinearity: comparing methods for disentangling the effects of habitat loss and fragmentation. Landscape Ecology 24:1271.

Smith, E. and M. Wishnie. 2000. Conservation and subsistence in small-scale societies. Annual Review of Anthropology 29:493-524.

Smyth, C., J. Schieck, S. Boutin, and S. Wasel. 2005. Influence of stand size on pattern of live trees in mixedwood landscapes following wildfire. The Forestry Chronicle 81:125-132.

Soroka, S. 2002. Issue attributes and agenda-setting by media, the public, and policymakers in Canada. International Journal of Public Opinion Research 14:264-285.

Soule, M. 1986. Conservation biology and the "real world". Pages 1-12 in Conservation Biology: the Science of Scarcity and Diversity. Sinauer Associates Inc., Sunderland, MA.

Soule, M. 1987. Where do we go from here? Pages 175-183 in Viable Populations for Conservation, edited by M. Soule. Cambridge University Press, Cambridge, UK.

Soule, M. 2013. The "new conservation." Conservation Biology 27:895-897.

Sovada, M., R. Woodward, and L. Igl. 2009. Historical range, current distribution, and conservation status of the swift fox, Vulpes velox, in North America. The Canadian Field-Naturalist 123:346-367.

Spangenberg, J. and J. Settele. 2010. Precisely incorrect? Monetising the value of ecosystem services. Ecological Complexity 7:327-337.

SPI. 2018. Species in the Balance: Partnering on Tools and Incentives for Recovering Canadian Species at Risk. Sustainable Prosperity Institute, online report: https://institute.smartprosperity.ca/.

Srivastava, D. and M. Vellend. 2005. Biodiversity-ecosystem function research: is it relevant to conservation? Annual Review of Ecology, Evolution, and Systematics 36:267-294.

Steel, B., P. List, D. Lach, and B. Shindler. 2004. The role of scientists in the environmental policy process: a case study from the American west. Environmental Science & Policy 7:1-13.

Steenweg, R., M. Hebblewhite, R. Kays, J. Ahumada, and J. Fisher. 2017. Scaling-up camera traps: monitoring the planet's biodiversity with networks of remote sensors. Frontiers in Ecology and the Environment 15:26-34.

Stekel, P. 1995. Did Chief Seattle Really Say, “The Earth Does Not Belong to Man; Man Belongs to the Earth?” Wild West Magazine, online article: www.historynet.com/chief-seattle.

Stelfox, J. 1995. Relationships Between Stand Age, Stand Structure, and Biodiversity in Aspen Mixedwood Forests in Alberta. Alberta Environmental Centre, Vegreville, AB.

Stevens, S., D. Page, and D. Prescott. 2008. Habitat Suitability Index for the Northern Leopard Frog in Alberta: Model Derivation and Validation. Government of Alberta, Edmonton, AB.

Stewart, D., Z. Underwood, F. Rahel, and A. Walters. 2018. The effectiveness of surrogate taxa to conserve freshwater biodiversity. Conservation Biology 32:183-194.

Stirling, I. and C. Parkinson. 2006. Possible effects of climate warming on selected populations of polar bears (Ursus maritimus) in the Canadian Arctic. Arctic 59:261-275.

Stradling, D. and R. Stradling. 2008. Perceptions of the burning river: deindustrialization and Cleveland’s Cuyahoga River. Environmental History 13:515-535.

Stralberg, D., X. Wang, M. Parisien, F. Robinne, and P. Solymos. 2018. Wildfire-mediated vegetation change in boreal forests of Alberta, Canada. Ecosphere 9:e02156.

Strong, W. and L. Hills. 2005. Late-glacial and Holocene palaeovegetation zonal reconstruction for central and north-central North America. Journal of Biogeography 32:1043-1062.

Suffling, R., M. Evans, and A. Perera. 2003. Presettlement forest in southern Ontario: ecosystems measured through a cultural prism. The Forestry Chronicle 79:485-501.

Sullivan, M. 2003. Active management of walleye fisheries in Alberta: dilemmas of managing recovering fisheries. North American Journal of Fisheries Management 23:1343-1358.

Sumners, W. and O. Archibold. 2007. Exotic plant species in the southern boreal forest of Saskatchewan. Forest Ecology and Management 251:156-163.

Surovell, T. and B. Grund. 2012. The associational critique of Quaternary overkill and why it is largely irrelevant to the extinction debate. American Antiquity 77:672-687.

Sutter, G. and R. Brigham. 1998. Avifaunal and habitat changes resulting from conversion of native prairie to crested wheat grass: patterns at songbird community and species levels. Canadian Journal of Zoology 76:869-875.

Suttle, K., M. Thomsen, and M. Power. 2007. Species interactions reverse grassland responses to changing climate. Science 315:640-642.

Svancara, L., M. Scott, C. Groves, R. Noss, and R. Pressey. 2005. Policy-driven versus evidence-based conservation: a review of political targets and biological needs. BioScience 55:989-995.

Swanson, M., J. Franklin, R. Beschta, C. Crisafulli, and D. DellaSala. 2011. The forgotten stage of forest succession: early-successional ecosystems on forest sites. Frontiers in Ecology and the Environment 9:117-125.

Swift, T. and S. Hannon. 2010. Critical thresholds associated with habitat loss: a review of the concepts, evidence, and applications. Biological Reviews 85:35-53.

Tanner, M. 1997. The evolution of environmental management in Canada. London Journal of Canadian Studies 13:1-11.

Tasker, J. 2017. Amid trans mountain uncertainty, pro-pipeline Indigenous peoples make a pitch for development. CBC, online news article: www.cbc.ca/news/politics/pro-pipeline-indigenous-people-trans-mountain-1.4253470.

Taylor, M., P. McLoughlin, and F. Messier. 2008. Sex-selective harvesting of polar bears (Ursus maritimus). Wildlife Biology 14:52-60.

Tear, T., P. Kareiva, P. Angermeier, P. Comer, and B. Czech. 2005. How much is enough? The recurrent problem of setting measurable objectives in conservation. BioScience 55:835-849.

Tedesco, D. 2015. American foundations in the Great Bear Rainforest: philanthro-capitalism, governmentality, and democracy. Geoforum 65:12-24.

Ter Steege, H., N. Pitman, D. Sabatier, C. Baraloto, and R. Salomao. 2013. Hyperdominance in the Amazonian tree flora. Science 342:1243092.

Thomas, C., A. Cameron, R. Green, M. Bakkenes, and L. Beaumont. 2004. Extinction risk from climate change. Nature 427:145-148.

Thompson, B. 2011. Planning for implementation: landscape-level restoration planning in an agricultural setting. Restoration Ecology 19:5-13.

Thorpe, J., N. Henderson, and J. Vandall. 2006. Exotic Tree Species as an Adaptation Option to Climate Change in the Western Canadian Boreal Forest. Summary Document No. 08-03. Prairie Adaptation Research Collaborative, Saskatoon, SK.

Thurfjell, H., S. Ciuti, and M. Boyce. 2014. Applications of step-selection functions in ecology and conservation. Movement Ecology 2:4.

Tigner, J., E. Bayne, and S. Boutin. 2015. American marten respond to seismic lines in northern Canada at two spatial scales. PloS One 10:e0118720.

Timoney, K. 1996. The logging of a world heritage site: Wood Buffalo National Park, Canada. The Forestry Chronicle 72:485-490.

Timoney, K. 2003. The changing disturbance regime of the boreal forest of the Canadian Prairie Provinces. The Forestry Chronicle 79:502-516.

Tindall, D. 2003. Social values and the contingent nature of public opinion and attitudes about forests. The Forestry Chronicle 79:692-705.

Tingley, M., E. Darling, and D. Wilcove. 2014. Fine-and coarse-filter conservation strategies in a time of climate change. Annals of the New York Academy of Sciences 1322:92-109.

Tittler, R., C. Messier, and A. Fall. 2012. Concentrating anthropogenic disturbance to balance ecological and economic values: applications to forest management. Ecological Applications 22:1268-1277.

Tittler, R., C. Messier, and R. Goodman 2016. Triad forest management: local fix or global solution. Pages 33-45 in Ecological Forest Management Handbook, edited by G. Larocque. CRC Press, Boca Raton US.

Tori, G., S. McLeod, K. McKnight, T. Moorman, and F. Reid. 2002. Wetland conservation and Ducks Unlimited: Real world approaches to multispecies management. Waterbirds 25:115-121.

Traill, L., B. Brook, R. Frankham, and C. Bradshaw. 2010. Pragmatic population viability targets in a rapidly changing world. Biological Conservation 143:28-34.

Trombulak, S. and C. Frissell. 2000. Review of ecological effects of roads on terrestrial and aquatic communities. Conservation Biology 14:18-30.

Troy, A. and K. Bagstad. 2009. Estimating Ecosystem Services Values in Southern Ontario. Spatial Informatics Group, Pleasanton, CA.

Tucker, M., K. Bohning-Gaese, W. Fagan, J. Fryxell, and B. Van Moorter. 2018. Moving in the Anthropocene: global reductions in terrestrial mammalian movements. Science 359:466-469.

Turner, C. 2013. The War on Science: Muzzled Scientists and Wilful Blindness in Stephen Harper's Canada. Greystone Books, Vancouver, BC.

Turner, N. and F. Berkes. 2006. Coming to understanding: developing conservation through incremental learning in the Pacific Northwest. Human Ecology 34:495-513.

Turner, R. and G. Daily. 2008. The ecosystem services framework and natural capital conservation. Environmental and Resource Economics 39:25-35.

Turnhout, E., C. Waterton, K. Neves, and M. Buizer. 2013. Rethinking biodiversity: from goods and services to "living with." Conservation Letters 6:154-161.

UN. 1992. The Rio Declaration on Environment and Development. The United Nations Conference on Environment and Development, Rio de Janeiro, Brazil.

UN. 2010. Decision Adopted by the Conference of the Parties to the Convention on Biological Diversity at its Tenth Meeting. Tenth Conference of the Parties, Nagoya, Japan.

UN. 2015. Paris Agreement, Framework Convention on Climate Change. 21st Conference of the Parties, Paris, France.

UN. 2022. News Release: Nations Adopt Four Goals, 23 Targets for 2030 in Landmark UN Biodiversity Agreement. UN Convention on Biological Diversity, Montreal, QC.

Urban, M., J. Tewksbury, and K. Sheldon. 2012. On a collision course: competition and dispersal differences create no-analogue communities and cause extinctions during climate change. Proceedings of the Royal Society of London B: Biological Sciences 279:2072-2080.

Usher, P. 2000. Traditional ecological knowledge in environmental assessment and management. Arctic 53:183-193.

Valiela, D. and C. Baldwin. 2007. Dealing with Mining Legacy—Some Canadian Approaches. Lawson Lundell LLP, Vancouver, BC.

van der Hoek, Y., A. Wilson, R. Renfrew, J. Walsh, and P. Rodewald. 2015. Regional variability in extinction thresholds for forest birds in the north-eastern United States: an examination of potential drivers using long-term breeding bird atlas datasets. Diversity and Distributions 21:686-697.

Van Vuuren, D., J. Edmonds, M. Kainuma, K. Riahi, and A. Thomson. 2011. The representative concentration pathways: an overview. Climatic Change 109:5-31.

VanNijnatten, D. 1999. Participation and environmental policy in Canada and the United States: Trends over time. Policy Studies Journal 27:267-287.

Veitch, A. 2001. An unusual record of a white-tailed deer, Odocoileus virginianus, in the Northwest Territories. Canadian Field Naturalist 115:172-175.

Venier, L., I. Thompson, R. Fleming, J. Malcolm, and I. Aubin. 2014. Effects of natural resource development on the terrestrial biodiversity of Canadian boreal forests. Environmental Reviews 22:457-490.

Venter, O., N. Brodeur, L. Nemiroff, B. Belland, and I. Dolinsek. 2006. Threats to endangered species in Canada. BioScience 56:903-910.

Vinebrooke, R., D. Schindler, D. Findlay, M. Turner, and M. Paterson. 2003. Trophic dependence of ecosystem resistance and species compensation in experimentally acidified Lake 302S (Canada). Ecosystems 6:0101-0113.

Virkkala, R. and A. Rajasarkka. 2007. Uneven regional distribution of protected areas in Finland: consequences for boreal forest bird populations. Biological Conservation 134:361-371.

Vitt, P., P. Belmaric, R. Book, and M. Curran. 2016. Assisted migration as a climate change adaptation strategy: lessons from restoration and plant reintroductions. Israel Journal of Plant Sciences 63:250-261.

Vitt, P., K. Havens, and O. Hoegh-Guldberg. 2009. Assisted migration: part of an integrated conservation strategy. Trends in Ecology & Evolution 24:473-474.

Volney, W. and R. Fleming. 2000. Climate change and impacts of boreal forest insects. Agriculture, Ecosystems and Environment 82:283-294.

Vucetich, J., M. Nelson, and J. Bruskotter. 2017. Conservation triage falls short because conservation is not like emergency medicine. Frontiers in Ecology and Evolution 5:45.

Waddington, J., P. Morris, N. Kettridge, G. Granath, and D. Thompson. 2015. Hydrological feedbacks in northern peatlands. Ecohydrology 8:113-127.

Wagner, R., J. Flynn, C. Mertz, P. Slovic, and R. Gregory. 1998. Acceptable practices in Ontario's forests: differences between the public and forestry professionals. New Forests 16:139-154.

Wallace, S., B. Turris, J. Driscoll, K. Bodtker, and B. Mose. 2015. Canada's Pacific groundfish trawl habitat agreement: a global first in an ecosystem approach to bottom trawl impacts. Marine Policy 60:240-248.

Walters, C. 2007. Is adaptive management helping to solve fisheries problems? Ambio 36:304-307.

Walters, C. and R. Hilborn. 1978. Ecological optimization and adaptive management. Annual Review of Ecology and Systematics 9:157-188.

Walther, G., A. Roques, P. Hulme, M. Sykes, and P. Pyvsek. 2009. Alien species in a warmer world: risks and opportunities. Trends in Ecology & Evolution 24:686-693.

Wang, X., G. Huang, and B. Baetz. 2016a. Dynamically-downscaled probabilistic projections of precipitation changes: A Canadian case study. Environmental Research 148:86-101.

Wang, T., A. Hamann, D. Spittlehouse, and C. Carroll. 2016b. Locally downscaled and spatially customizable climate data for historical and future periods for North America. PLoS One 11:e0156720.

Wang, X., D. Thompson, G. Marshall, C. Tymstra, and R. Carr. 2015. Increasing frequency of extreme fire weather in Canada with climate change. Climatic Change 130:573-586.

Wang, Y., Pedersen, J., Macdonald, S., Nielsen, S. and J. Zhang. 2019. Experimental test of assisted migration for conservation of locally range-restricted plants in Alberta, Canada. Global Ecology and Conservation 17:e00572.

Waples, R. 1998. Evolutionarily significant units, distinct population segments, and the Endangered Species Act: reply to Pennock and Dimmick. Conservation Biology 12:718-721.

Ward, C. and T. Erdle. 2015. Evaluation of forest management strategies based on Triad zoning. The Forestry Chronicle 91:40-51.

Warren, F. and D. Lemmen. 2014. Canada in a Changing Climate: Sector Perspectives on Impacts and Adaptation. Natural Resources Canada, Ottawa, ON.

Waters, S. 2010. Swift fox Vulpes velox reintroductions: a review of release protocols. International Zoo Yearbook 44:173-182.

WCED (World Commission on Environment and Development). 1987. Our Common Future. Oxford University Press, Oxford, UK.

Weber, M. and W. Adamowicz. 2002. Tradable land-use rights for cumulative environmental effects management. Canadian Public Policy 28:581-595.

Weber, J. and C. Word. 2001. The communication process as evaluative context: what do nonscientists hear when scientists speak? BioScience 51:487-495.

Weir, J., S. Fuhlendorf, D. Engle, T. Bidwell, and D. Cummings. 2013. Patch Burning: Integrating Fire and Grazing to Promote Heterogeneity. Oklahoma State University, Stillwater, OK.

Wessels, K., S. Freitag, and A. Van Jaarsveld. 1999. The use of land facets as biodiversity surrogates during reserve selection at a local scale. Biological Conservation 89:21-38.

West, J., S. Julius, P. Kareiva, C. Enquist, and J. Lawler. 2009. US natural resources and climate change: concepts and approaches for management adaptation. Environmental Management 44:1001-1021.

Whittaker, R., S. Levin, and R. Root. 1973. Niche, habitat, and ecotope. The American Naturalist 107:321-338.

Wiens, J., D. Ackerly, A. Allen, B. Anacker, and L. Buckley. 2010. Niche conservatism as an emerging principle in ecology and conservation biology. Ecology Letters 13:1310-1324.

Wiens, T., B. Dale, M. Boyce, and G. Kershaw. 2008. Three way k-fold cross-validation of resource selection functions. Ecological Modelling 212:244-255.

Wiersma, Y. 2005. Environmental benchmarks vs. ecological benchmarks for assessment and monitoring in Canada: is there a difference? Environmental Monitoring and Assessment 100:1-9.

Wiersma, Y. and T. Nudds. 2006. Conservation targets for viable species assemblages in Canada: are percentage targets appropriate? Biodiversity Conservation 15:4555-4567.

Wiersma, Y. and C. Simonson. 2010. Canadian national parks as islands: investigating the role of landscape pattern and human population in species loss. Park Science 27:70-77.

Wiersma, Y. and D. Sleep. 2018. The effect of target setting on conservation in Canada's boreal: what is the right amount of area to protect? Biodiversity and Conservation 27:733-748.

Williams, B. 2011. Passive and active adaptive management: approaches and an example. Journal of Environmental Management 92:1371-1378.

Williams, B., R. Szaro, and C. Shapiro. 2009. Adaptive Management: The U.S. Department of the Interior Technical Guide. Adaptive Management Working Group, U.S. Department of the Interior, Washington, DC.

Williams, H., J. McMurray, T. Kurz, and F. Lambert. 2015. Network analysis reveals open forums and echo chambers in social media discussions of climate change. Global Environmental Change 32:126-138.

Williams, J. and S. Jackson. 2007. Novel climates, no-analog communities, and ecological surprises. Frontiers in Ecology and the Environment 5:475-482.

Williams, R. and P. O'Hara. 2010. Modelling ship strike risk to fin, humpback and killer whales in British Columbia, Canada. Journal of Cetacean Research and Management 11:1-8.

Williamson, T., H. Hesseln, and M. Johnston. 2012. Adaptive capacity deficits and adaptive capacity of economic systems in climate change vulnerability assessment. Forest Policy and Economics 15:160-166.

Willig, M., D. Kaufman, and R. Stevens. 2003. Latitudinal gradients of biodiversity: pattern, process, scale, and synthesis. Annual Review of Ecology, Evolution, and Systematics 34:273-309.

Willis, K. and G. MacDonald. 2011. Long-term ecological records and their relevance to climate change predictions for a warmer world. Annual Review of Ecology, Evolution, and Systematics 42:267-287.

Willis, K. and R. Whittaker. 2002. Species diversity—scale matters. Science 295:1245-1248.

Wilson, H., L. Joseph, A. Moore, and H. Possingham. 2011. When should we save the most endangered species? Ecology Letters 14:886-890.

Wilson, J. 1998. Talk and Log: Wilderness Politics in British Columbia, 1965–1996. UBC Press, Vancouver, BC.

Wilson, K. and E. Law. 2016. Ethics of conservation triage. Frontiers in Ecology and Evolution 4:112.

Wilson, K., M. Westphal, H. Possingham, and J. Elith. 2005. Sensitivity of conservation planning to different approaches to using predicted species distribution data. Biological Conservation 122:99-112.

Winterhalder, B. and F. Lu. 1997. A forager-resource population ecology model and implications for indigenous conservation. Conservation Biology 11:1354-1364.

Woodroffe, R. and J. Ginsberg. 1998. Edge effects and the extinction of populations inside protected areas. Science 280:2126-2128.

Wood, S., G. Tanner, and B. Richardson. 2010. Whatever happened to Canadian environmental law? Ecology Law Quarterly 37:981-1040.

Wotton, B., M. Flannigan, and G. Marshall. 2017. Potential climate change impacts on fire intensity and key wildfire suppression thresholds in Canada. Environmental Research Letters 12:095003.

Wulder, M. 2011. Can Satellites Help Monitor Biodiversity More Effectively? Canadian Forest Service, online report: http://cfs.nrcan.gc.ca/publications?id=32949.

Wurtzebach, Z. and C. Schultz. 2016. Measuring ecological integrity: history, practical applications, and research opportunities. BioScience 66:446-457.

WWF. 2010. 100 Million Hectares, 10 Per Cent of Canada, 1 Generation. How WWF Put Conservation on the Map. WWF, Toronto, ON.

WWF. 2018. Transforming Business. WWF website, retrieved March 1, 2018: www.worldwildlife.org/initiatives/transforming-business.

Wyatt, S., J. Fortier, D. Natcher, M. Smith, and M. Hebert. 2013. Collaboration between Aboriginal peoples and the Canadian forest sector: A typology of arrangements for establishing control and determining benefits of forestlands. Journal of Environmental Management 115:21-31.

Y2YCI. 2014. The Yellowstone to Yukon Vision: Progress and Possibility. Yellowstone to Yukon Conservation Initiative, online report: https://y2y.net.

Yaffee, S. 1999. Three faces of ecosystem management. Conservation Biology 13:713-725.

Yeboah, D., H. Chen, and S. Kingston. 2016. Tree species richness decreases while species evenness increases with disturbance frequency in a natural boreal forest landscape. Ecology and Evolution 6:842-850.

Yeoman, B. 2014. Billions to none. Audubon Magazine 116:28-33.

Yodzis, P. 1989. Introduction to Theoretical Ecology. Harper and Rowe, New York, NY.

Zhang, J., W. Kissling, and F. He. 2013. Local forest structure, climate and human disturbance determine regional distribution of boreal bird species richness in Alberta, Canada. Journal of Biogeography 40:1131-1142.

Zhang, J., S. Nielsen, J. Stolar, Y. Chen, and W. Thuiller. 2015. Gains and losses of plant species and phylogenetic diversity for a northern high-latitude region. Diversity and Distributions 21:1441-1454.