EXPLORING THE SOCIAL-ECOLOGICAL RESILIENCE OF FOREST ECOSYSTEM SERVICES

by

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ABSTRACT

Natural disturbance is predicted to increase in Canadian forests as the climate continues to change. This will trigger an increased variability, and therefore uncertainty, in the supply of ecosystem services from forests. I used social-ecological systems theory to develop a forest management approach that recognizes and incorporates spatial and temporal dynamics. Social-ecological approaches integrate the role of people in ecosystems. This approach focuses on the maintenance of social and ecological resilience to change as the main management objective. I developed a structured framework that examines a resource system's social and ecological dynamics and the supply of provisioning and regulatory ecosystem services. Systems modelling was used to capture the overall behaviour of forest resources in the Cranbrook timber supply area and as a foundation for developing scenarios that identified a range of future ecosystem conditions. I then used spatio-temporal simulation models to capture a range of future environmental and social conditions, including climate change. Natural disturbance was implemented to reflect historic variability. The supply of ecosystem services, under all scenarios, oscillated through time driven by the interaction of natural disturbance and forest management, making a constant supply unattainable. A sustainable timber supply is possible if harvest levels are lower than those currently prevailing; suitable habitat for grizzly bears can be sustained at high or low levels depending on road densities and access rules. A social-ecological approach is well suited to understanding drivers of change, sources of uncertainty, and in managing the supply of ecosystem services from dynamic ecosystems.

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CHAPTER 1

General Introduction

Background

The intent of forest management in British Columbia (BC) is to plan for a consistent supply of timber while providing for other ecosystem services (Province of BC 2007). The current mountain pine beetle (*Dendroctonus ponderosae*; MPB) outbreak and the anticipated changes in climate (IPCC 2007) undermine the ability of resource professionals and policy makers to maintain expected levels of services using current forest management approaches (Carpenter 2003, Folke et al. 2004, Walker and Meyers 2004, Adger et al. 2005, MA 2005, Hobbs et al. 2006, Williamson et al. 2009). This anticipated shortfall provides the impetus to develop a new approach that integrates landscape dynamics and ecological resilience into forest management.

Landscape dynamics, the maintenance of ecological values, and timber production all have social and ecological components. Thus, we require a new approach for forest planning that captures those two, sometimes competing elements. Social-ecological systems theory can provide the foundation for such an approach. A social-ecological system is characterized by resilience, adaptability and transformability (Walker et al. 2004). When managing such a system the goal becomes not only to provide a sustainable supply of ecosystem services, but also to explicitly account for the social and ecological dynamics that may beset them.

Social-ecological systems theory views resource management as the integration of natural and human dynamics and the capacity of the system to respond to change while maintaining its defining functions and structures (Holling 1973, Gunderson and Holling 2002, Drever et al. 2006). In this context, resilience represents the ability of ecosystems to maintain their defining features and processes following natural and human disturbance. Adaptability is the capacity of the ecosystems, species and human actors in the system to adjust to both ecological and social change. Transformability is the ability of the system to transform from its current configuration to a different configuration. The transformation is triggered when the resilience and adaptability to disturbance is overcome and a new system emerges, organized around a different set of defining structures, functions and controls (Walker et al. 2004).

In this chapter, I introduce the foundational ideas for developing a social-ecological systems approach to forest resource planning. This includes an overview of complex systems theory, and the related concepts of resilience and adaptive cycles. Following an overview of the conventional resource management paradigm I provide an introduction to an alternative social-ecological systems-based approach. I end the chapter with a description of the thesis goal and objectives.

Complex Systems Theory

Until the early 20th century, science viewed the universe as a machine, governed by the basic laws of determinism, with man as a separate entity on the outside. The theories of relativity and quantum mechanics were seen as corrections to classical theory. People moved from being impartial observers, to being part of the description of nature (Prigogine 1986). At the same time, the second law of thermodynamics was being reconciled with the disorder of entropy. There were questions of how creative processes were to be reconciled at higher orders of organization, such as biological evolution. The second law was valid for a closed system; however, it became evident that there was an “entropy flow” in open non-equilibrium systems, where there was interaction with the environment (Capra 1996). The entropy of internal systems decreases, as energy is consumed from an external environment and the entropy of the external environment increases with the conversion of energy. Over time, this results in the destruction of pre-existing order, as sub-systems move towards equilibrium. This awareness led to a new model: one based on the world being complex; that emphasized the duality of destruction and creativity inherent in natural, “open” systems (Prigogine 1986).

This open systems view was buttressed by contemporary experiments in chemistry and physics. They concluded that when physical or chemical systems are far from thermodynamic equilibrium, unexpected structures and patterns emerge. Prigogine (1986) termed these as “dissipative structures”: when simple chemical reactions acquire complex, “emergent” behaviour. These observations prompted future studies of complex systems to integrate a system's history, its elements, relations, evolution, and overall behaviour.

Complex systems theory has its roots in physics, ecology and Gestalt psychology and a common theme is the duality of reductionism and holism (Capra 1996). Complex systems are typically defined as systems the behaviour of which is not fully explained by an understanding of their parts (Gallagher and Appenzeller 1999). Deconstructing a system and analysing its constituents destroys the organized relations between the parts. Typically, a complex system’s parts are coupled in a non-linear way. They are characterized by feedback loops; they are open, with their boundaries difficult to determine; they have a history, where past system states influence future states; they may be nested, with higher levels of organization made up of constituent sub-levels; and they have emergent properties. Analysis and synthesis describe a system at different scales. Analysis assumes that a system is a bounded unit that can be described, whereas, synthesis views not the parts, but their interactions and context (Ritchey 1996).

Richardson et al. (2001) state that one of the shortcomings of systems theory is that it tries to be a theory of everything. The result is that it becomes unbounded and open ended, making it difficult to conduct analysis and to operationalize. Solutions and methodologies to describe complex systems are themselves emergent. They are dependent on the frames of reference of the actors involved, and the context of the problem being investigated. What is developed is not easily transportable to other, even similar, systems (Richardson et al. 2001).

Complex systems can be better understood through analogy. Hypotheses can be formulated about the fundamental principles that a system must satisfy in order to perform. It is not a description of its components, but a description of its actions. The interactions of a spruce budworm (*Choristoneura fumiferana*) cycle and moose (*Alces alces*) browsing provide an example of a complex system. A spruce (*Picea* sp.) forest may be defoliated by spruce budworm, killing the trees. As the forest begins to recover, trembling aspen (*Populus tremuloides*) and paper birch (*Betula papyrifera*) tend to dominate the forest, representing a different ecological state. Moose selectively browse the deciduous trees, suppress their growth and regeneration and thereby facilitate the re-establishment of conifers; this leads to a mature spruce forest again (Ludwig et al.

1978). The spruce and the birch-aspen forest are two alternative ecological regimes. The forest oscillates between each regime based on the actions and adaptability of spruce budworm and moose. In contrast to conventional analysis -- that focuses on only one component of a system -- complex systems theory provides a perspective to describe and analyse the dynamics of a system.

A social-ecological system is a type of complex system with defining structures and functions: it is made up of ecological, social and economic domains. Viewing the ecological processes and human activities -- responsible for future forest condition and ecosystem services -- as a linked social and ecological system, provides insights into their complex dynamics. This approach assists in understanding the consequences of actions and sources of uncertainty (Carpenter et al. 2001, Peterson et al. 2003a). A complex system can be characterized by a set of state variables that reflect the status of its elements, or structures of interest, and its processes, or functions (Walker et al. 2004). A complex system has the following properties (after Snowden and Boone 2007, and Campbell et al. 2009):

- parts (e.g., trees, animals) and processes (e.g., growth, succession, disturbance, species dispersion), that interact with one another and their environment over multiple scales of time and space;
- feedbacks that can be amplifying (positive), or dampening (negative);
- non-linearity, where minor changes can produce disproportionally large unpredictable changes;
- memory, where past system states influence the current and future configurations of the system (e.g., propagules); and

- a pattern of the global system, that emerges solely from the numerous interactions among lower level components, without being guided by an external higher level source.

A description of a social-ecological system not only includes the components and dynamics within domains, but also their interactions, feedbacks, memory and states. It is best portrayed by assessing the attributes of resilience, adaptability and transformability (Walker et al. 2004).

Resilience

Ecological resilience theory emerged from complex systems theory in the early 1970s (Holling 1973). The concept of resilience incorporated aspects of complex systems theory including, a systems view of ecosystems, an understanding of the relationships, as well as the parts of ecosystems, nested levels of organization, and feedbacks between processes and scales through space and time. In the ecological literature the concept of resilience has multiple meanings (Drever 2006, Brand and Jax 2007). However, within the context of social-ecological systems theory, it is considered to be the ability of a system to maintain its defining structures, functions, identity and processes (Carpenter et al. 2001). As an example, forest-dependent wildlife have evolved to exist in a landscape with specific structural and functional characteristics, and are maintained by natural ecosystem dynamics (Bunnell 1995, Wong et al. 2003). The system's resilience would be dependent on the maintenance of those ecosystem dynamics, and the resulting pattern and age of vegetation. In a human exploited forest, the resilience of the system includes the capacity of the forested ecosystems to maintain their defining structures and processes, despite the additional disturbances prescribed by forest management. Resilience has three defining properties: (1) the amount of change a system can go through and still retain the same controls, structure and function, (2) the capacity of the system to self-organize around new controls, and (3) the degree to which the system can learn and adapt (Carpenter et al. 2001).

In contrast to ecological resilience (Holling 1973, Carpenter et al. 2005), an engineer's view of resilience is characterized as being the capacity of a system to return to its equilibrium state after a perturbation (Peterson 1998), such as a rubber band returning to its original form after being stretched. This interpretation of resilience is focused on the state of the system. Alternatively, ecological resilience focuses on the processes of change and the maintenance of relationships, incorporating natural fluctuations in ecological expression. Ecosystems can exhibit engineered resilience, being resistant to change, as well as the flexibility to change typified by ecological resilience. Ecosystems may exhibit a particular state or 'ecological regime' as characterized by a consistent set of traits (Lewontin 1969); however, there is always some level of variation across space and time. From a complex systems perspective, a system that tends towards a stable configuration is characterized as having an "attractor", and this part of the state space behaves as a "basin of attraction". The system could be continually in motion within its basin of attraction driven by, for example, stand level ecosystem dynamics. These ecological and engineering views of resilience reflect differences in world views. An engineer's resilience is a more classic scientific view and would see the world and its natural systems in equilibrium, and any departure would be undesirable and temporary and controllable. An ecological resilience perspective would recognize the relationships, dynamics and fluctuations of a system as being part of its overall behaviour.

Ecosystems do not remain in one particular state through time, but through succession, disturbance, and a variety of dynamic landscape processes potentially assume different states. This concept of "multiple stable states" is central to resilience theory with ecosystems potentially transitioning between several basins of attraction (Gunderson 2000, Scheffer et al. 2001, Beisner et al. 2003, Walker et al. 2004). Resilience considers the amount of disturbance or perturbation that a system can withstand before changing between these alternative stable states (Carpenter et al. 2001). For example, a wildlife system could be characterized by a dominant ungulate, and the extent, pattern and structure of old forest. A combination of those conditions would describe the system's state at a particular time. Specific human or ecological processes may maintain a system in a particular state or configuration. For example, a system dominated by woodland caribou (*Rangifer tarandus caribou*) may be converted to an alternative moose state, by either wildfire, timber extraction or road construction, and then may be maintained by industrial forestry activities for a period of time. Alternately, the system could convert back to a caribou dominated state.

Adaptive Cycle

A key component of social-ecological systems theory is the concept of the adaptive cycle (Figure 1-1; Gunderson and Holling 2002). Systems are driven by forces of growth and reproduction, for example, an increase in forest biomass as trees grow. Similarly, systems are propelled by novelty such as wildfires that disturb forests and facilitate the growth of different tree species. An adaptive cycle suggests that systems do not necessarily tend towards equilibrium, but instead cycle dynamically. Gunderson and Holling (2002) state that systems go through four stages: growth and exploitation (r), conservation (K), release or collapse (α), and renewal and reorganization (Ω). The first two stages of growth and conservation are slow and are termed the front loop, and it is during these first stages where efficiency is maximized. The second two stages, or back loop, are typically much quicker, as the system reorganizes and provides the foundation

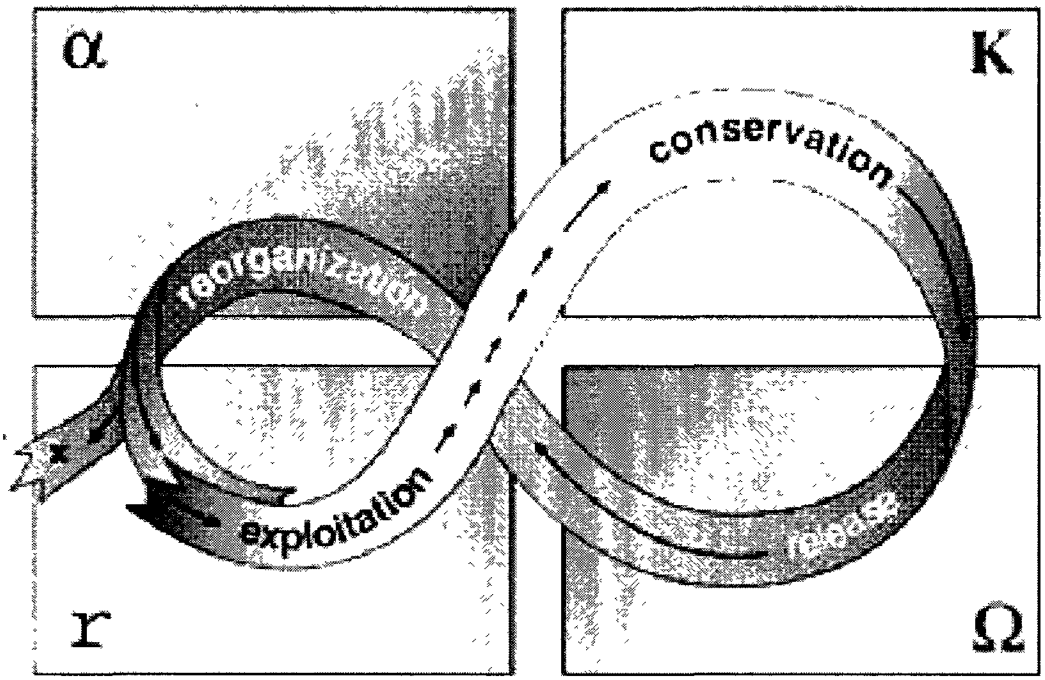

Figure 1-1. Stylized drawing showing four phases of the adaptive cycle: exploitation or growth, conservation, release, and reorganization (r, K, α, Ω). The arrows between the phases represent time; longer arrows represent the shorter periods during the conservation, release, and exploitation phases. The short arrows represent the longer period between the exploitation and conservation phases. Novelty entering the system is represented by the break in the loop in the reorganization phase. The loop can exit during reorganization, if the system undergoes a regime shift. (Gunderson and Holling 2002).

for system novelty to enter the cycle. The adaptive cycle reflects the apparent long-term stability of a system, as well as the periodic catastrophic upheavals that occur. The constant renewal and reorganization of a system is a reflection of its resilience through its capacity to adapt, change, and yet maintain processes that lead to its renewal (Gunderson and Holling 2002).

The disturbance ecology of the boreal forest provides an example of the four stages of an adaptive cycle. Forest growth and succession reflect the front loop of the adaptive cycle, where biomass accumulates. Here, the system is highly resilient initially and is capable of absorbing a range of disturbances, both at the stand and landscape scale, without becoming fundamentally altered. However, as the forest matures and becomes more highly connected, there is an increase in biomass; it becomes more vulnerable to being drastically altered by landscape disturbance, although the forest may be able to rebound from stand-scale disturbance (Drever et al. 2006). Under the back loop -- or release and reorganization phase -- external forces, such as defoliating insects or wildfire, overwhelm the organization of the forest. This leads to a loss of mature canopy cover, and as the forest reorganizes novelty enters the system with different vegetation and wildlife occupying the landscape (Drever et al. 2006).

Conventional Approach to Forest Management

Modern forest management aims to provide a relatively constant supply of ecosystem services, as they produce economic benefit, ecosystem health and human well being. Such ecosystem services include: provisioning (timber, non-timber products, wildlife and a clean supply of drinking water), cultural and regulatory services (biological conservation and hydrological balance) (MA 2005). However, conventional approaches to forest management are challenged with the task of maintaining historic levels of service (MA 2005, Puettmann et al. 2009). A major contributing factor to this challenge is the assumption of environmental and economic certainty that underlies current practices. Little flexibility is provided in management plans to accommodate unforeseen events, be they catastrophic natural disturbance, or the results of incomplete knowledge of how forestry activities may impact ecosystems (Hunter 1990, Holling and Meffe 1996, Utzig and MacDonald 2000, Robinson 2004).

Forest practices have led to a homogenization of forest composition, pattern and structure, compared with historical conditions that were shaped by natural ecological processes (McRae et al. 2001). Further, harvesting has become more common than fire in many Canadian forests (CCFM 2010). The reduction of natural ecosystem processes, such as fire and regeneration, has led to an increased susceptibility to catastrophic disturbance, and has contributed to a declining capacity of forests to provide a consistent supply of ecosystem services (Bergeron et al. 2002, Kuuluvainen 2002, Drever et al. 2006).

Social-Ecological Systems Approach to Management

Resource management based in social-ecological systems theory has emerged as an alternative that addresses the challenges faced by the conventional approach in dealing with social and ecological dynamics (Gunderson and Holling 2002, Walker et al. 2004). Developing different ways of managing resources is becoming urgent, particularly as global systems become stressed by factors such as climate change and the disruption of natural ecological systems due to human activities (MA 2005). A social-ecological approach integrates people with nature (Berkes and Folke 1998). This approach recognizes the values ascribed by society to the forested environment, and the physical and biological properties of the system that provide those values. Within a social-ecological approach, the focus is on recognising and maintaining forest composition, pattern and structure, as well as ecosystem processes.

There are similarities between a social-ecological approach to forest management and one that uses historic natural disturbance to guide forest management. Both approaches aim to maintain or restore historic composition, pattern and structural diversity of forests (Attiwill 1994, Bunnell 1995, Bergeron and Harvey 1997, Angelstam 1998, Seymour et al. 2002, Drever et al. 2006, Klenk et al. 2009). However, a social-ecological approach is also concerned with the resilience and adaptability of a system to disturbance. Ecosystem processes and their dynamics determine the temporal availability, and overall capacity of the system to supply ecosystem services. Further, a social-ecological approach considers the interplay between local and regional scales that create social and ecological heterogeneity. This recognition of social and biological diversity, across space and time, promotes the long-term persistence of ecosystem services across landscapes (Peterson 2002, Walker et al. 2002, Drever et al. 2006, RA 2007a).

The concepts of resilience and adaptive cycles are used in the social-ecological systems approach to interpret the sustainability of ecosystem services. Through this approach, services can be better aligned with dynamic ecosystem processes, and still be maintained across large areas and through time. A social-ecological system links the social and ecological dimensions of ecosystem services. In this context, the resilience of the social-ecological system becomes the capacity of forested ecosystems to maintain their defining structures and processes, despite forest harvesting activities. The resilience of a social-ecological system is also related to the ability of people to adapt to new ecological conditions and still maintain their livelihoods. In a social-ecological approach there is more attention to flexibility, both biologically, through the encouragement of diversity to better enable forest renewal and reorganization after disturbance, and socially, so that expectations of what level of services should be available are more aligned with the local ecosystem dynamics. Together, these reflect the adaptability of the social-ecological system. Transformation of the system is embodied in the understanding of longer-term ecosystem dynamics, accepting that what was present in the past is not necessarily going to be in the future, such as forests converting to grasslands under climate change.

Implementing a Social-Ecological Approach to Forest Management

A social-ecological perspective has been applied to the management of other resource systems, including a lake system in Wisconsin, U.S.A. (Peterson et al. 2003b), rangeland in Australia (Walker et al. 2002), and forests in Florida, U.S.A. (Peterson 2002). In these examples of "resilience management", there is a focus on describing ecosystems, their dynamics at relevant scales, their dependent social systems, and how human action and natural resource use interact. Scenario modelling (Peterson et al. 2003a) is used to explore a range of plausible futures to assist in understanding the relationships between the system's components and the implications of a particular management action, or inaction, relative to social-ecological outcomes. Through this process, the adaptability of the social-ecological system emerges as the people involved in a social-ecological assessment identify the vulnerabilities of the system to various human or natural surprises and prescribe interventions to encourage system states more favourable to the continued supply of ecosystem services.

Thesis Objectives

The goal of my thesis research is to develop and apply methods for evaluating social-ecological systems. I will focus on the supply of ecosystem services from dynamic forested resource systems and address the following objectives:

1) Develop a framework for describing dynamic forested resource systems as social-ecological systems. The framework will include a description of a system's resilience to natural and human disturbance events, and its adaptability to ecological and social change.
2) Define a set of scenarios that describe a range of possible future social and ecological conditions for a resource system that will serve as a case study.
3) Implement a set of simulation experiments to model the social and ecological processes of a resource system.
4) Quantify and compare the ecosystem services resulting from each defined scenario.

The Cranbrook timber supply area in southeastern BC will be used as the land base for demonstrating the framework and scenario analysis. Across that area, timber supply and coarse- and fine-filter biodiversity will be assessed. The response variable for timber supply will be the volume of wood available for harvest through time, and growing stock. I will use the area of old forest by ecosystem grouping and the area of natal habitat for grizzly bear (*Ursus arctos*) as an index of coarse- and fine-filter biodiversity, respectively.

Organization of Thesis

The thesis is organized into four chapters. This chapter introduced the objectives of the thesis, and provides an overview of the foundational concepts and theoretical context for developing a social-ecological systems framework to describe forest management. Chapter two addresses the first and second thesis objectives, where I develop a framework and accompanying scenarios for describing forest ecosystems and management as a social-ecological system. In the third chapter I pursue the third and fourth objectives, through an analysis and assessment of a set of scenarios that embody the main forces and uncertainties of ecological and social change in the Cranbrook study area. In the final chapter I synthesize the findings of the research and discuss the utility of the approach for evaluating a social-ecological system, its resilience to natural and human disturbance, and the uncertainty associated with the supply of ecosystem services.

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CHAPTER 2

Scenario composition: considering natural resource management as a social-ecological system

Introduction

There has been a recent increase in extreme natural disturbance events: flooding in Pakistan (Khan et al. 2010); bleaching of coral reefs in Australia (Hoegh-Guldberg 1999); and insect outbreaks in western Canada (Eng et al. 2005, 2006, Safranyik and Wilson 2006). These disturbance events threaten agriculture, marine and forest resources, as well as put human lives and infrastructure at risk (Carpenter 2003, Walker and Meyers 2004, Adger et al. 2005, Folke et al. 2004, MA 2005, Hobbs et al. 2006, Williamson et al. 2009). Large-scale disturbance events are expected to increase as the climate continues to change (Emanuel 2005, Hoegh-Guldberg et al. 2007, IPCC 2007, Williamson et al. 2009). Conventional management paradigms are unprepared to deal with the impacts of large-scale catastrophes relative to the provisioning of ecosystem services (Folke et al. 2004). Even before the onset of recent extensive natural disturbance events, there were questions regarding the ecological sustainability of conventional approaches to management (MA 2005), particularly how suitable they were to managing dynamic ecosystems (Gunderson and Holling 2002, Drever et al. 2006, Lindenmayer et al. 2008, Puettmann et al. 2009).

Conventional approaches to resource management implement strategies intended to maximize the return of specific commodities, while striving to minimize impacts on non-commodity ecosystem services (Holling and Meffe 1996, Scheffer et al. 2001, Ludwig et al. 2005). There is an assumption of certainty underlying current resource management. The premise is that any future disturbance to a resource is controllable, and ecosystems can be manipulated to maintain a consistent supply of commodities (Holling and Meffe

1996). Based on this assumption, management practices tend to homogenize the spatial arrangement of ecosystems and their dynamics to maximize the extraction of resources. In pursuit of production efficiencies, this course of action ends in the decline of functional, compositional and structural diversity, compromising the capacity of ecosystems to recover from perturbation (Chapin et al. 1996, Pastor et al. 1998, Folke et al. 2004).

Resource management plans typically forecast a single sequence of events and do not evaluate a range of possible futures (Holling and Meffe 1996, Peterson et al. 2003). As they are driven by economic and social pressure to provide as much immediate benefit as possible, these plans discount the future and the disruptions caused by natural disturbance (Holling and Meffe 1996). Resource management regimes have also become socially entrenched, and are supported by institutional bodies and regulations that are largely maladaptive (Gunderson 1999, Westley 2002).

Social-ecological systems theory provides a foundation to develop an alternative approach to manage resources. A social-ecological systems perspective, based in systems theory (Forrester 1961, Gallagher and Appenzeller 1999, Meadows 2008), views natural resources within a larger social and ecological context; it describes people and the environment as a linked "resource system" (Walker et al. 2004, RA 2007). Social-ecological resource planning accepts and anticipates future natural disturbance. In an effort to ensure a long-term supply of ecosystem services, interventions are prescribed that increase system resilience and adaptability (Walker et al. 2002, Peterson et al. 2003, Walker et al. 2004). Therefore, a social-ecological framework is well suited to manage for an unpredictable future (Gunderson and Holling 2002, Walker et al. 2002, Berkes et al. 2003, Walker et al. 2004, Folke et al. 2004, RA 2007).

Scenario planning has emerged as an effective technique to operationalize social-ecological systems theory (Peterson et al. 2003, MA 2005, Carpenter et al. 2006). Using this technique, a range of scenarios is composed that consider the breadth of possible social and ecological change. Any single scenario is not a prediction; however, it illustrates how specific events may influence the future. Scenario planning exercises are appropriate when developing management strategies for dynamic ecosystems with uncertain future trajectories (Gunderson and Holling 2002, Peterson et al. 2003, Walker et al. 2002, Carpenter et. al. 2006, Campbell et al. 2009).

There are two main objectives of this chapter. The first is to develop a framework for describing dynamic forested resource systems as social-ecological systems. The approach is generic and applicable to a range of resource management contexts; however, for brevity this paper will focus on the ecology and management of boreal and montane forests. The second objective is to define a set of scenarios that describe a range of possible future social and ecological conditions of a resource system that will serve as a case study.

The chapter is divided into four sections. The first section outlines the rationale for developing a social-ecological framework for describing resource systems. The second part provides background on social-ecological system theory and its application to the management of forests. The third section introduces the social-ecological framework for describing resource systems. Based on the social-ecological framework, the methods for defining a set of scenarios that capture a range of social and ecological drivers of change are also presented. The fourth part of the chapter discusses a forest management unit in southeastern BC that is undergoing an extreme MPB outbreak event, as an example of the social-ecological approach. I conclude with a discussion of the utility of the social-ecological framework and the scenario planning approach.

1. Forest Dynamics and Management

1.1 Dynamics

The capacity of a forest to withstand a large-scale disturbance event and effectively recover to its pre-disturbance state is strengthened by the functional redundancy and response diversity of its ecosystems (Peterson et al. 1998, Bergeron et al. 2002, Diaz et al. 2003, Elmqvist et al. 2003, Drever et al. 2006). Ecosystems have a range of species that fill the same role, such as burrowers and nitrogen fixers (Brown and Heske 1990, Marcot et al. 2002). The loss of any one species is not considered to have an overwhelming impact on the ecosystem due to functional redundancy (Folke et al. 2004). For example, in the Columbia Basin of BC, where agriculture has replaced native grasslands, American badgers (*Taxidea taxus*), burrowing owls (*Athene cunicularia*) and two species of ground squirrels (*Urocitellus columbianus, U. washingtoni*) have been extirpated and the ecological role of burrowers has been replaced by different species of gophers (Geomyidae), mice (*Mus musculus*) and voles (*Microtus californicus, M. canicaudus*) (Marcot et al. 2002). However, even though species may fill the same functional role, they undoubtedly respond to environmental change differently. The interaction of these various responses across scales provides redundancy thereby increasing the probability for functional roles persisting post-disturbance or as a landscape's climate shifts. This variability is termed response diversity (Elmqvist et al. 2003, Campbell et al. 2009). Further, management that encourages landscape complexity -- multi-scale composition, pattern and structural diversity -- can help buffer ecosystems against the spread of disturbance (Turner et al. 1998, Puettmann et al. 2009).

In most conifer forests, a large portion of ecological complexity is the product of the forest's natural disturbance regime -- the rate, extent, severity of disturbance, and the post-disturbance biological legacies (Pickett and White 1985, Puettmann et al. 2009). For any particular disturbance regime, these characteristics are variable, leading to the diversity and spatial arrangement of habitats seen across a landscape (Burton et al. 2003, Turner et al. 2003). The main natural disturbance agents of the boreal forest are fire, insects and windstorms (Suffling and Perera 2004), but other events, such as tree diseases, are being increasingly recognized as important drivers of landscape composition and heterogeneity (Bergeron 1998). Natural disturbance regimes vary from frequent small-scale low-intensity, to infrequent large-scale high-intensity events that release and reallocate ecosystem resources, resulting in dramatic changes in landscape composition and structure (Shiel and Burslem 2003, Lavigne and Gunnell 2006).

The mechanisms that influence various disturbance agents differ. For example, a landscape's fire regime is a function of weather conditions, ignition agents, fuel availability, and fire suppression (Schoennagel et al. 2004). Fire, driven by weather, transforms vegetation and forest community structure. Shifts in weather will influence post-disturbance vegetation establishment, thereby altering community structure (Paine et al. 1998, Jasinski and Payette 2005, McIntire et al. 2005, Johnstone and Chapin 2006). On the other hand, the mountain pine beetle (*Dendroctonus ponderosae*; MPB), currently impacting large areas of the North American cordillera (Eng et al. 2005, 2006, Safranyik and Wilson 2006), has a disturbance regime that is a function of the availability of host trees, primarily mature lodgepole pine (*Pinus contorta*), and of weather conditions; mild winters facilitate brood survival (Taylor and Carroll 2004).

There is geographic variation in the frequency and extent of disturbance across the forested regions of Canada. For example, the Montane Cordillera ecozone, the mountainous area of western North America, is highly variable, with a fire cycle ranging from 30 to 300 years or more (Wong et al. 2003). East of the Montane Cordillera, the disturbance rate varies from 50 to 100 years in the west, to 100 to 300 years in Ontario and Quebec (Bergeron et al. 2001). In eastern Labrador the fire return interval is much longer, at 500 years (Foster 1983).

In addition to being variable in space, there is extensive temporal variability in the disturbance regimes of the boreal forest (Johnson et al. 1998, Bergeron et al. 2001, Daniels et al. 2007, Krawchuck et al. 2009, Meyn et al. 2009). The concept of the "range of natural variability" (RONV) has been promoted as a tool to more fully characterize disturbance regimes (Cissel et al. 1999, Landres et al. 1999, Haeussler and Kneeshaw 2003). By using the RONV, the focus shifts toward understanding the full dynamics of the system, not simply the central tendency of some attribute (Haeussler and Kneeshaw 2003). However, there remain two core challenges to this approach when applied to forested systems. First, there is no consensus on the time period for describing regional stability of disturbance regimes in unlogged forests. Prior to European contact, a traditional system of land use practices established by indigenous peoples probably played a significant role in determining landscape condition (Suffling and Perera 2004). This system was supplanted by an industrial system, where the influence of humans more dramatically altered the natural disturbance regime (Suffling and Perera 2004). The second challenge is that RONV does not account for disturbance events that are subject to cyclical forces that modify the characteristics of a regime (Hunter 1988, Weir et al. 2000). For example, fire frequency in western North America is influenced by the Pacific Decadal Oscillation (PDO) and El Niño/La Niña ocean temperature oscillations. This causes the regional disturbance regime to temporally vary in frequency, extent and severity, and at times trigger large regional fires (Turner et al. 1998, Daniels et al. 2007, Morgan et al. 2008). This flux throughout the Holocene has led some researchers to conclude that, for any landscape in the boreal forest, there is no single characteristic disturbance regime (Bergeron et al. 1998).

Large episodic natural disturbance events play a critical role in forest complexity. Through the resetting of successional cycles, these events contribute to the spatial and compositional diversity of forests (Turner et al. 1998). Further, they can provide a resource pulse (Holt 2008) that can, for a period of time, increase the abundance of certain vegetation communities or wildlife food sources. For example, grizzly bear (*Ursus arctos)* populations in southeastern BC increased with an expanded availability of huckleberries (*Vaccinium* spp.): the product of large historic wildfires (McLellan and Hovey 1995).

Climate change is already altering disturbance regimes, and it is anticipated to have an even greater influence on the future dynamics of the boreal and Cordilleran forest (Hobbs et al. 2006, Williamson et al. 2009). Current influences include large outbreaks of MPB (Taylor and Carroll 2004, Eng et al. 2005, 2006, Safranyik and Wilson 2006) and rapid increases of Dothistroma needle blight (*Mycosphaerella pini*) (Woods et al. 2006). A number of studies have demonstrated that climate change will influence the frequency and extent of wildfire, with increases of up to 100% in expected annual area burned (Wotton and Flannigan 1993, Stocks et al. 1998, Flannigan et al. 2005, Li et al. 2000, Nitschke and Innes 2008, Krawchuk et al. 2009). These studies conclude that the chance of fire will increase, due to climate change-driven increases in fire season length and fire weather severity. They also suggest that fires will be more volatile and difficult to control, as they shift from a mean behavioural regime of surface fires with torching to one more frequently characterized by crown fires. Further, due to the longer fire season and drier conditions in some forests, these larger more frequent and severe fires will undermine the capacity of forest managers to conserve biodiversity, protect the habitat of species at risk, and ensure a sustainable supply of timber for harvest. Understanding how fire season length and drought incidence will be affected by climate change gives insights into how fire regimes may shift spatially.

1.2 Management

A fully regulated or so-called “normal” forest has been the underlying objective for traditional forest management, where forests are managed to be homogenous stands with a uniform age class structure (Puettmann et al. 2009). Managed forests are assumed to be at equilibrium: where extreme perturbation is uncommon and undesirable; where risk and uncertainty, related to disturbance events, is minimal (Gunderson 2000). This premise of a perfectly engineered and controlled forest continues to influence modern forestry practices, although biodiversity, habitat, and areas with aesthetic and recreation value are included as constraints to timber harvesting (Perry 1998, Bourgeois 2008, Puettmann et al. 2009).

Under conventional forest management, natural disturbance regimes are altered, primarily through the suppression of fires, in an effort to increase the supply of mature trees for harvest and to meet conservation objectives. So prevalent is this approach, that the area disturbed by harvesting has exceeded that of fire in some jurisdictions (Figure 2-1: CCFM 2010). As well, under forest management there is a shift in the frequency of disturbance, from a 50 - 500 year return interval for fire, to a 40 - 100 year return interval for timber harvesting (McRae et al. 2001). By minimizing natural ecosystem processes, such as fire and regeneration, there has been an increased susceptibility to catastrophic disturbance (Bergeron et al. 2002, Kuuluvainen 2002, Drever et al. 2006). For example, land use changes and direct fire exclusion increases fuels, and decreases the gap in structure between tree crown and forest floor, and as a result, increases the risk of high-severity fires in mixed-severity landscapes (Arno et al. 2000). Large infrequent events are a dominant component of the disturbance regime (Stocks et al. 2002), and are important generators of diversity (Burton et al. 2008). However, Ryan (2000) suggests that though fires may have become less frequent, they are now more severe, due to the accumulation of dead fuel and increased density of the understory, overwhelming the capacity of the forest to recover. Further, in the absence of fire, large areas of forest have become older and more susceptible to insect attacks, such as, the current MPB attack in BC (Taylor and Carrol 2004, Eng et al. 2005, 2006, Safranyik and Wilson 2006).

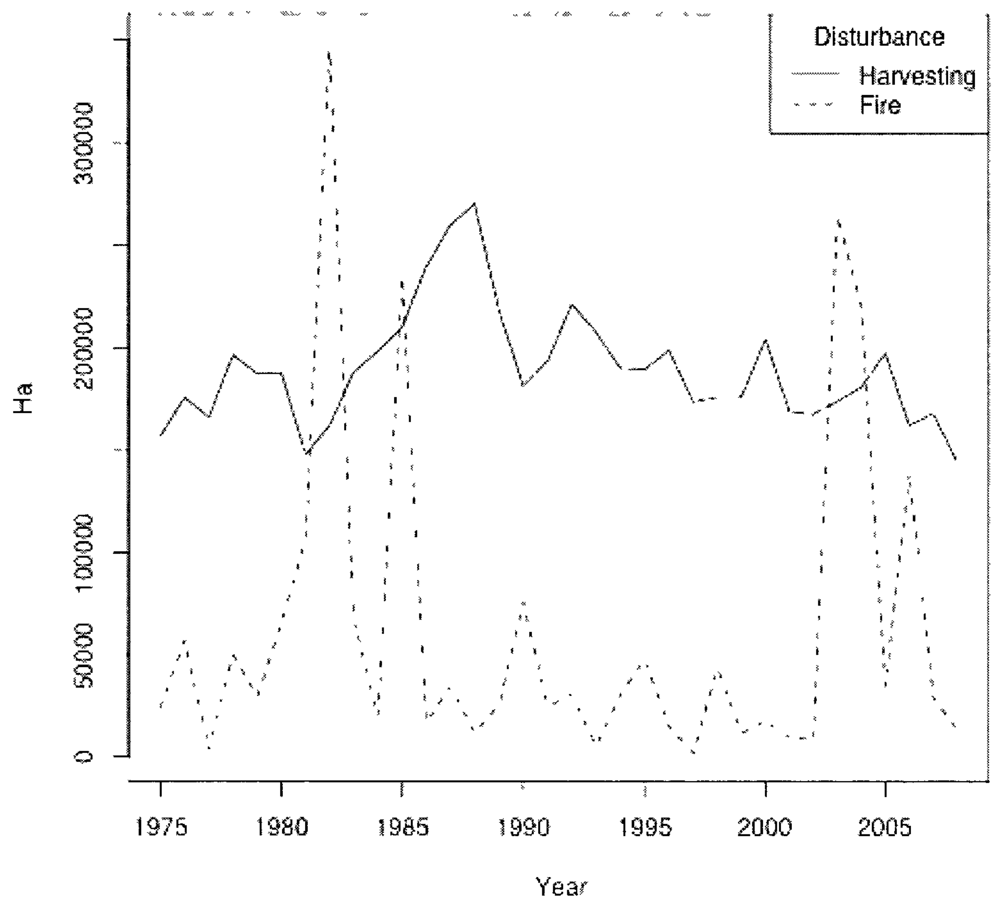

Figure 2-1. Area disturbed by fire and harvesting in British Columbia between 1975 and 2008 (CCFM 2010).

The spatial extent, frequency, temporal variability and legacies of disturbance all shift when forests are extensively managed. The result is a homogenization of the structure, pattern and composition of forests (Pastor et al. 1998, Buddle et al. 2006, Bergeron 2001, McRae et al. 2001, Kuuluvainen 2002, Lindenmayer and McCarthy 2002). Wildfires in the boreal forest create a range of patch sizes, varying from many small, to a few very large openings. Further, there are multiple pathways that a forest could take when recovering from a disturbance, which contributes to forest complexity (Turner et al. 1998, Lecomte et al. 2006). In contrast, forest management typically harvests uniformly sized areas, does not produce the periodic extensive openings that remain after large fires, tends to replant with monocultures, and leaves a smaller range of species and far less biomass on the site relative to post-fire legacies (Angelstam 1998, Elmqvist et al. 2003, Drever et al. 2006, Puettmann et al. 2009). Overall, this spatial and temporal homogenization has been implicated in the loss of forest function, response diversity, productivity and the abundance of some wildlife species (Peterson et al. 1998, Elmqvist et al. 2003, Drever et al. 2006, Campbell et al. 2009).

Emulating natural disturbance, where past natural disturbance is used as a template for forest management, has been suggested as one solution to address the spatial homogenization of forests (Hunter 1993, Attiwill 1994, Bunnell 1995, Bergeron and Harvey 1997, Angelstam 1998, Seymour et al. 2002, Drever et al. 2006). Under a natural disturbance based approach to management, the frequency, size, shapes and residual structure of natural disturbance events are mimicked to maintain the structure and patterns on the landscape that are consistent with historic RONV (Hunter 1993, Attiwill 1994, Kneeshaw et al. 2000, McRae et al. 2001, Bergeron et al. 2002, Haeussler and Kneeshaw 2003, Drever et al. 2006).

There are, however, several issues that compromise the implementation of natural disturbance based forestry management. For example, it is difficult to emulate large complex processes, such as natural disturbances (James and Norton 2002), where fire frequency and size are extremely variable within a landscape (Gill and McCarthy 1998). As well, there is little agreement on what would constitute a historical disturbance regime (Suffling and Perera 2004). Interpreting past disturbance is confounded by recent disturbance erasing evidence of former events (Morgan 1994). Further, emulating large fires may be difficult to accept by society. Overall, a natural disturbance based approach would result in a loss of timber supply (Binkley 1997). In response to these criticisms, the "triad" approach to forest management has been proposed (Hunter 1990, Seymour and Hunter 1992, Messier and Kneeshaw 1999). Essentially, it zones the forest into areas of intensive forestry, ecological reserves, and areas designated for multiple use; the matrix of the triad is managed according to natural disturbance based principles (Lindenmyer et al. 2006). However, with neither of these approaches extensively tested, there remain questions as to their efficacy to produce the complexity generated by natural processes (Buddle et al. 2006).

Along with the normal forest method, there are two long-term temporal issues that natural disturbance based and triad approaches do not address effectively. The first is the shifts in landscape processes historically observed and anticipated with climate change that undermine the ability for managers to infer future disturbance regimes (Hunter 1988, Bergeron et al. 1998, Emanuel 2005, Williamson et al. 2009). In many forest management regimes, the periodicity of disturbance has been truncated by using a single reconstruction of disturbance history as the basis for estimating timber losses from natural disturbance. It is common for potential losses to the forest, due to fire, insects or pathogens, to be based on the last 50 years of forest inventory records (BCMFR 2007). Limiting forest management to recent time periods ignores longer term dynamics driven by processes such as ocean-atmospheric oscillations, even though the signature of the PDO is evident in dendrochronological records of tree ring scars caused by fire and insect outbreaks (Bergeron et al. 1999, Weir et al. 2000, Daniels et al. 2007, Morgan et al. 2008). Using recent time periods serves the assumption that the recent past is the most suitable predictor of the future. In response to the limitations of using a short time frame for assigning management rate and extent of harvest, there has been greater effort to reconstruct longer term disturbance regime dynamics (Daniels et al. 2007). Despite these efforts, forest management plans predominantly prescribe a relatively consistent range of fire size and frequency to inform rates of cut and size of harvesting units. Assuming that an area's disturbance history can be described with a single prescription is consistent with the expectation that a managed forest should be homogenous and is at equilibrium, despite the research to the contrary.

At the stand scale, the equilibrium assumption made by forest managers is evident in estimates of timber yield. The volume of timber that is harvested in the short term is dependent on the long-term availability of trees for future harvest. However, it is assumed that replanted trees are to grow in a predictable manner, with only as much mortality or disturbance-based losses as prescribed by the recent disturbance history (Puettmann et al. 2009). As a result, forest management can end up prescribing rates of harvest that are based on only one temporal disturbance and growth pattern from recent history, instead of incorporating the temporal dynamics and variability that are evident in the dendrochronological record.

The second temporal issue that undermines the normal and natural disturbance based planning approaches is the incidence and impact of large infrequent disturbances. Current management, in whatever form, focuses on one prescriptive future as the basis for rate, extent and pattern of harvesting. Typically, there is no flexibility to integrate large episodic events driven by climate oscillations that are an important component of forest diversity (Hunter et al. 1988, Turner et al. 1998). Further, if systems with large frequent disturbances, such as fire, are also subject to forest harvesting, there is the danger that they will be subjected to compounded perturbations, with the rate of disturbance being faster than the rate of recovery (Paine et al. 1998). Salvage activities can also compromise medium- and long-term timber supply, and thus the sustainability goals of forest management plans (Spittlehouse and Stewart 2003, Coates et al. 2006, Lindenmayer et al. 2008).

In addition to ecological variability leading to uncertainty in the future supply of ecosystem services, there is shifting public interest in what the forest should and can provide. Historically, timber for harvest was adequate; now there are social expectations of forests for recreation, wildlife and old forest preservation (CCFM 2003) that undermine long-term expectations regarding how much of the forest can be dedicated to timber extraction. Meeting social interests requires innovative compromises to ensure a continued unfettered access to forest for harvest (Price et al. 2009, Canadian Boreal Forest Agreement 2010). Further, there is growing interest in non-timber products, and in preserving forests as carbon stores, a climate change mitigation strategy (Millar et al. 2007).

1.3 Management Systems

Optimal models of resource use, applied by forest managers to support decision making, depend on certainty and consistency of the future supply of a resource (Holling and Meffe 1996, Scheffer et al. 2000, Ludwig et al. 2005, Morgan et al. 2007). Under an optimal approach, the real system is represented by a predictive model; however, if the system behaviour deviates too much from the model representation, the predictions fail (Walker et al. 2002). Operationally, management related activities are designed and implemented around the optimal solution, and an expectation is established of a consistent timber flow to support a labour force and processing facility. Ecological systems have become dominated by an engineering paradigm, and managed as a system with narrow operating objectives, such as annual allowable cut. The optimal strategy of forest management may be effective for highly managed or controlled systems, where external sources of variability can be controlled; however, this approach has been found to be flawed, due to the challenge of trying to control poorly understood complex dynamic systems (Holling and Meffe 1996, Scheffer et al. 2000, Ludwig et al. 2005).

To administer ecological systems there has been a tendency towards a “command and control” style of management. This consolidation of power and capacity into fewer more centralized organizations erodes the flexibility of local managers to adapt to location-specific and novel conditions (Holling and Meffe 1996, Bodin and Norberg 2005). With more hierarchical management systems it is a challenge for managers to modify their activities in response to unique socio-economic circumstances and extreme events. Large infrequent disturbances stress agencies responsible for forest management and can lead to errors that have impacts on forest function (Westley 2002, Foster and Orwig 2006, Lindenmayer and Noss 2006). Change in management style only seems to occur when a resource manager is faced with an ecosystem crisis (Gunderson 1999, Westley 2002). In response, adaptive management, a strategy to experiment, gain knowledge and then develop appropriate strategies in response (Walters 1986), is promoted as a method for improving the flexibility and effectiveness of forest management.

Analysis of large-scale disturbances, such as the Mount St. Helens volcanic eruption of 1980 (Franklin and MacMahon 2000) and the Yellowstone National Park forest fires of 1988 (Turner et al. 2003), provides insights that can help managers prepare for future large-scale events. Many researchers now are advocating for a revolution in resource management that moves away from a centralized, reactionary paradigm to a more adaptive, autonomous, and proactive approach (Holling 1986, Gunderson and Holling 2002, Turner et al. 2003, Walker et al. 2004). This type of management is more compatible with dynamic ecosystems and promotes communication within organizations allowing for a more efficient response to change.

2. Resource Management as a Social-Ecological System

Ecosystem dynamics are recognized as a central component of management under a social-ecological perspective. Strategies are developed that incorporate the system's variability and uncertainty, and provide a range of options on managing for the future (Carpenter et. al. 2001, Gunderson and Holling 2002, Walker et al. 2004, RA 2007, Campbell et al. 2009). The approach recognizes the connection between people and their interest in ecosystem services, and the dynamic and complex environment that provides them. Social-ecological systems are characterized by their resilience, adaptability and transformability (Walker et al. 2004).

2.1 Resilience

Resilience has three defining properties: (1) the amount of change a system can go through and still retain the same controls, structure and function; (2) the capacity of the system to self-organize around new controls; and (3) the degree to which the system can learn and adapt (Carpenter et al. 2001). The resilience of a managed forest is the capacity of ecosystems to maintain their defining structures and processes, despite the additional disturbances prescribed by forest management, and to recover to a previous condition if disturbed (Carpenter et al. 2001). Managing for resilience involves understanding how disturbance forces interact with the forest, and managing the condition of the forest so that it can better withstand change and continue to provide ecosystem services.

A resilient forest is not necessarily one that is stagnant in one particular state, but in a cycle of disturbance, renewal and growth. The metaphor of "adaptive cycles" is used to describe the phases of such a cycling system: growth, conservation, release and reorganization (Figure 1-1; Gunderson and Holling 2002). Consider a forested landscape as it cycles through the four phases. As a forest becomes established there is a rapid proliferation in the number of seedlings and an accumulation of biomass -- the growth phase. Once at stand maturity, the system is maintained until the accumulated biomass, or capital locked up in old trees, and the system's carrying capacity is reached -- the conservation stage. A disturbance event, such as a fire, causes the system to free up the stored capital in the system -- the release phase. From this state of chaos early successional species compete for dominance -- the reorganization phase. The resilience of the system to disruption is strongest during the reorganization or growth phase, where the system is more capable of recovery to its previous condition. The system is least flexible and most vulnerable during the conservation phase. It takes a significantly longer time to regenerate forest during this phase, therefore, the resilience of the system is weakest (Gunderson and Holling 2002).

The adaptive cycle can be considered at multiple scales: forest stand to landscape, to the regional scale consisting of multiple landscapes (Holling 1992, Gunderson and Holling 2002). The hierarchical model of ecosystem dynamics is termed a panarchy. Fast disturbance processes at smaller scales generate spatial diversity in ecological structure that provide a degree of buffering against an extreme future disturbance event; for example, the variability in stand response to insect and disease, limiting their spread at larger landscape or regional scales (Gunderson and Holling 2002). Larger scale slower processes create a cross-scale feedback. These conserve or destroy biological legacies such as seed banks or species migration, that determine how ecosystems reorganize after disturbance (Campbell et al. 2009). As a result, resilience of a forest is grounded in ecological processes that are localized and fast, as well as in processes that are slow and occur at larger scales (Carpenter and Levitt 1991, Levin 1992, Gunderson and Holling 2002).

Managing for resilience entails a number of activities, which could include (after Campbell et al. 2009):

- introducing fire to ecosystems where it has been excluded to re-establish natural processes;
- managing for a diversity of stand ages and compositions to reduce exposure to future disease and insect outbreaks (Woods et al. 2005, Campbell et al. 2008);
- varying the size and shape of forest cut blocks and leave areas to buffer against windthrow disturbance (Kimmins 2004);
- varying the mix of species when replanting to limit homogenous stands that could be vulnerable to future disturbance; and
- planting genotypes that are more resistant to pests, disease or are more suitable for an emerging climate (Millar et al. 2007, O'Neil et al. 2008).

In summary, maintaining resilience is achieved by prescribing management strategies that either support the system in a desired condition, or reduce the resilience of a system that is in an undesirable configuration, in an effort to encourage a state that provides a preferred suite of ecological services (Walker et al. 2004, Bennet et al. 2005, Carpenter et al. 2005, Cumming et al. 2005, RA 2007).

2.2 Adaptability

The adaptability of social-ecological systems is dependant on how well the system responds in both the social and ecological domain. An example of the latter is the capacity of the system to adapt to the change in forest composition as a response to a disturbance event, such as spruce recruitment and release following pine mortality from MPB (Coates et al. 2006). From a social perspective, adaptive capacity could refer to the ability of the forest industry to adjust its milling capacity to process a far greater proportion of beetle-killed timber (Byrne et al. 2006).

Anticipating uncertainty and adapting to change is an integral part of managing a social-ecological system (Gunderson and Holling 2002). Social adaptability is related to the flexibility of local management and the business networks, as well as the social assets such as education and skills of the workforce and availability of technology (Holling and Meffe 1996, Carpenter et al. 2001, Bodin and Norberg 2005, McAfee et al. 2010). Communities may change how they use forest products -- for example, switching from solid timber products to those intended to serve as bio-fuel (BCMFR 2007) -- or design flexible zoning strategies for industry and conservation (Rayfield et al. 2008). Further, it includes increasing the dialogue with interest groups to debate the appropriate level of risk to take towards resource extraction, and how these resources should be managed. Through interest group collaboration, adaptability can be enhanced by conducting pre-disturbance planning that identifies procedures and strategies in order to be prepared for a large-scale disturbance event (Lindenmayer et al. 2008). Other adaptation planning measures might include: increasing the representation of ecosystems in areas reserved from harvest; identifying and conserving areas that could be refugia from the effects of climate change (Rose and Burton 2009); or protecting biologically important landscape features (Pojar 2010).

2.3 Transformability

Resilience and adaptability are features of the same regime, whereas transformability is the process of a different regime becoming established. Specifically, transformability is the ability of a system to organize around a new set of defining structures, functions and controls (Walker et al. 2004). In the social domain, transformability is the capacity of the people, within a social-ecological system, to create a new system when the current system becomes unworkable (Walker et al. 2004). In an ecological system, transformability occurs when the reinforcing processes that maintain a system are overcome by slowly changing system dynamics, or by an acute disturbance. The system shifts and an alternative regime emerges (Gunderson 2000, Scheffer et al. 2001, Beisner et al. 2003, Walker et al. 2004). For example, an established landscape may alternate between an open forest and grassland, where established grasslands are maintained by the reinforcing processes of fire and herbivory (Starfield et al. 1993, Cumming et al. 1997). Alternatively, the open forest state may persist because shading limits grasses, which in turn limit the spread of fire (Walker 1989, Dublin et al. 1990).

During transformation the social and ecological domains interact. Commercial forestry, natural disturbance and the expansion of human settlement could alter the configuration and composition of the landscape by changing the use of the land, through conversion or degradation of ecosystems and habitat (MA 2005). These changes may be benign initially, but when a critical threshold is reached their cumulative impact may cause the system to reorganize into a different configuration; a landscape switches from being dominated by natural processes, to one maintained by extensive human management (Scheffer et al. 2001). A landscape may still be forested, but the pattern and structure of the forest has changed through silvicultural practices (Puettmann et al. 2009). The limit of a system's alternative states can be determined based on the historic fluctuation of its state variables, driven, for example, by the rates and extent of an area's disturbance regime. It is also necessary to consider the influence of past human activity on the landscape, such as the historic extent of grazing or settlement patterns.

Finally, transformation concepts are particularly relevant to forest management and can be incorporated into management plans. For example, in an area with a rapidly changing climate, accounting for transformation could mean relocating species or developing strategies that facilitate species migration such as north-south corridors (Millar et al. 2007, Pojar 2010).

2.4 Summary of Conventional vs. Social-Ecological Approaches to Forest Management

The central limitations of conventional approaches to forest management are the loss of complexity necessary to buffer forests against large-scale disturbance, and the lack of adaptation in dealing with shifting disturbance regimes. Through land conversion, fire suppression and forest harvesting, humans have altered disturbance regimes. The result is a decrease in the natural diversity, which has led to a spatial and temporal homogenization of forest pattern, composition and structure. Although large infrequent events play a role in a forest's disturbance regime, with the forest losing diversity there is the possibility that disturbance becomes more common and extreme (Bergeron et al. 2002, Kluuvuainen et al. 2002, Drever et al. 2006).

Optimization strategies lead to a loss of resilience due to the focus on one commodity and the blanket application of the same management regime across the landscape (Bodin and Norgerg 2005). The system becomes brittle, with no capacity to absorb unknowns because of the lack of variation and options (Gunderson 2000). A resilience based approach to forest management, one that considers adaptability and transformability, addresses these shortcomings by implementing alternative stand and landscape stewardship practices. These practices are designed to enhance the functional redundancy and response diversity of ecosystem processes, vegetation communities and wildlife, as a means of buffering against large-scale episodic disturbance, and to aid in post-disturbance recovery and reorganization (Campbell et al. 2009). Rather than using forecasting to decide on an optimal management strategy, a resilience based management approach would focus on the resilience of desirable system attributes, and use scenario planning techniques to consider a wide range of possible futures (Bennett et al. 2005).

Due to the complexity of natural processes, it would be impossible to perfectly emulate a natural disturbance regime; however, a social-ecological approach could implement a regime that would be adaptable to future regime shifts (Lindenmayer et al. 2008). By putting more emphasis on infrequent large-scale disturbance events, those potential system states at the boundary of possibility, the strategy anticipates the inevitable surprises and is more flexible in dealing with uncertainty. Table 2-1 summarizes the differences between conventional and a social-ecological approach to forest management.

An additional criticism of current management is the social dependency that develops, with management regimes and objectives becoming entrenched and centralized, limiting flexibility to deal with ecological change (Holling and Meffe 1996). Planning tends to focus on maximizing the supply of commodity services and optimizes for one preferred future, instead of managing for system diversity to increase ecological

Table 2-1. Comparison of conventional and social-ecologically based approaches to forest management.

		Conventional	Social-Ecological
Resilience	**Goal**	Long range sustainable yield	Increasing functional and response diversity
	Structure	Uniform age structure, harvest at culmination age	Age structure consistent with disturbance regime, variable harvest rotation age
	Pattern	Uniform blocks with some variation consistent with single historic snapshot	Variable, consistent with dendrochronological record, anticipate climate change influences
	Composition	Replant monoculture	Variable within and across areas replanted
	Ecological process	Fire suppression	Targeted fire suppression, increase in prescribed burning
Adaptability	**Goal**	Optimal harvest	Bet-hedging: anticipate future unknowns
	Key features	Limited conservation priority	High conservation priority
	Redundancy	Single ecological representation, low uniform post-harvest retention	Multiple ecological representation, variable levels of retention
	Connectivity	Limited	Multiple connections across scales
	Disturbance	Impacts averaged and assumed on an annual basis	Focus on variation and anticipate periodic large-scale salvage
Transformability	**Goal**	Managed landscape for no change	Manage transition among states
	Planning for change	Assume stable future, plan for single resource	Linked human-ecological system and cumulative effects of human activities, plan for multiple possible futures
	Extreme events	Assume consistent supply, if extreme event occurs then redo plans	Protocols for response to periodic large-scale events
	Relocation	Tree seed planting zones shifted when necessary	Facilitated migration of range of plants and animals when required

response to large-scale change. Given the inevitability of extreme disturbance, planning for landscape dynamics is an important adaptation strategy for ensuring a future supply of ecosystem services. Social-ecologically based planning would manage for dynamics, including an array of possible post-disturbance successional pathways.

Under a social-ecologically based management approach, post-harvest retention of trees would be variable and stands would be harvested at a range of ages, thereby varying rotation length across the forest (Bergeron et al. 1999, Burton et al. 1999, Seymour and Hunter 1999). These strategies would be aimed at maximizing diversity and ecological complexity across scales. Resilience of the forest to catastrophic disturbance is encouraged by managing for the suite of adaptive cycle phases.

3. A social-ecological systems approach to resource management

A general framework is required to implement the social-ecologically based management approach. The framework presented describes the components and relationships of a social-ecological system, issues of concern, and the social and ecological drivers of change (Cumming et al. 2005, Bennett et al. 2005, RA 2007). A set of possible futures, based on social and ecological variability and uncertainty, are then constructed to capture the behaviour of the social-ecological system and the mechanisms of change (Peterson et al. 2003).

The social-ecological approach to resource management is broken down into three main steps. The first step identifies the issue of concern, describes the current state of the system, its history and cross-scale interactions. The second step captures the overall behaviour of the system, including fast and slow drivers of change. Also noted in the second step are the critical thresholds between different system states, including the mechanisms that could lead the system to switch to either a different social-ecological state or into a different phase of its adaptive cycle. Scenario composition is the final step. Scenarios of possible future system configuration are used as a planning technique to capture information as part of a pre-disturbance strategy for dealing with extreme events.

3.1 Current Condition

The first step in developing the framework is to identify the issue of concern and the current conditions. The description includes the social-ecological system's ecosystems, ecological processes, dominant economic activities, the community vision of land use (i.e., plans) and governing institutions (RA 2007). Social-ecological systems can be the accumulation of numerous interactions among lower level processes. The resulting expression of the system emerges as a product of these interactions, and although its overall behaviour can be described, this description does not reveal the lower-level source phenomena. Further, many social-ecological systems are open to external influences, many of which are unknown. To effectively describe a social-ecological system, a sufficient level of detail must be applied to limit the parts, processes and scales that are under consideration. By defining the specific question being assessed and the spatio-temporal bounds of the system, a workable approach can be realized. For example, there may be adequate habitat for wildlife or area available for timber extraction presently, but there must be a sufficient supply through time given possible fluctuations. The time signature of these dynamics defines the temporal bounds of the focal system.

System interactions cross scales and form part of a description of current social conditions or forest composition and structure. For example, the age and species composition of a forest are a product of stand-level processes such as growth, succession and tree mortality. A forest is also influenced by larger-scale processes such as drought cycles or regional forest policy dictating allowable annual cuts. Socially, the human actors in a system can be split by scale into those that are primarily internal to the area, the local communities, and those that are primarily external, including corporate shareholders or government decision makers.

Social-ecological systems change through time. Documenting historic social, economic or ecological events helps to build understanding of the mechanisms that led to the current expression of the system and its variability. Finally, in the early stages of the project, it is important to initiate collaboration: early involvement of interest groups and decision makers increases the adoption of results from the assessment (Peterson et al. 2003, Fabricius et al. 2007, Berghofer et al. 2008).

3.2 Future conditions

When identifying future conditions, planning teams must recognize the overall behaviour of the system as well as the system's disturbance regime based on its past dynamics and how those dynamics could shift in the future. The goal is to gain an understanding of the controlling forces that have shaped the system and how they may interact, change and be influenced by other social and ecological events in the future. This requires a list of the main social and ecological drivers of the system and a description of the different ways in which they could interact in the future. Developing this list requires that one recognize and document the forces of change and cycles of change.

Forces of Change

The forces, or drivers, that act on the system can be categorized by the speed with which they act, their origin and whether or not they are the product of ecological processes or from human action. By analysing a system's historic behaviour, eliciting expert opinion or published accounts of similar systems, the forces that have been responsible for past change can be recorded (Nakicenovic and Swart 2000, MA 2005, IPCC 2007). The forces can be interpreted as stressing the system, either by moving it away from some idealized condition, or creating the conditions for a discrete event that will result in a loss of system integrity. The forces can have positive or negative (destabilizing or stabilizing) feedbacks. A phenomenon that involves a positive feedback is a nuclear chain reaction: once initiated it becomes self-reinforcing. A negative feedback dampens the oscillations of a system, such as lynx predation moderating the size of a snowshoe hare population.

Social-ecological systems typically have slow and fast drivers that shape the system through time (Carpenter and Levitt 1991, Levin 1991, Holling 1992). Slow drivers can be predictable, like the ecological drivers of soil development, or forest growth and succession. Alternatively, slow drivers like climate change may result in a great deal of uncertainty over time. Slow social drivers include population growth, land conversion, settlement expansion, or a developing road network. Fast drivers are usually a shock or disturbance event and the outcome is predictable, but often dramatic; examples of fast ecological drivers include a fire or insect outbreak. Fast social drivers may be flooding from the construction of a dam, or forest harvesting. Listing drivers, their social or ecological basis and whether they are fast or slow, is the first step in understanding an area's forces of change.

Resource management interventions also have the potential to act as slow drivers leading to unanticipated events, such as historic fire suppression contributing to fuel loading and an increase in wild fire severity (Arno et al. 2000). Other social drivers may be a change in technology that shifts which species of tree can be harvested, or a slow improvement in education or the composition of the local economy leading to a different set of social choices about how to manage the environment. The slow and fast system drivers do not act independently, but interact. For example, spruce budworm (*Choristoneura fumiferana*) populations are controlled by predators and the density of balsam fir (*Abies balsamea*). As the size and density of the trees increases, -- a slow driver -- the ability of birds to control budworms declines, and an outbreak (a fast driver) can be triggered (Holling 1973, Ludwig et al. 1978, Holling 1988).

Finally, when considering forces, it is important to acknowledge that there are interactions with phenomena outside of the focal system. Here, external forces act on the system, changing the behaviour of specific drivers. An example of such a phenomenon is the PDO-driven drought cycle, leading to large regional fires destroying open forest and resulting in expanded grasslands (Morgan et al. 2008).

Cycles of Change

The concepts of alternative states and adaptive cycles are useful for understanding and organizing the drivers, feedbacks, and cross-scale interactions of the system and provide insights into the position of the system within these cycles of change. Through an interpretation of slow controlling variables, the location of the social-ecological system relative to its current state or adaptive cycle can be deduced. The system could, for example, be close to a threshold that could signal an imminent shift. The position of the system would inform management to either implement strategies to encourage or discourage change, or that a transformation to a new phase or state is required. The slow variables that are directing the system towards a threshold provide a surrogate of system resilience (Carpenter et al. 2005). As an example, a threshold in a forest may be the extent of old pine that is susceptible to MPB, which may signal further outbreaks, and prompt management to focus on harvesting old susceptible pine in an effort to temper future outbreaks and the resulting loss of timber (Taylor and Carroll 2004). Alternatively, a system may not be cycling. It may have once cycled, but is now being held in a particular configuration through human management.

A description of the phases of the systems above and below the focal system helps identify the possibility of cross-scale interaction. For example, a forest composed of predominantly old pine stands may be susceptible to an epidemic landscape-scale outbreak when local stand-level MPB outbreaks start to spread, coalescing to a larger-scale event. Managing forests so that not all stands are at the same stage of the adaptive cycle enhances resilience to larger-scale events (Gunderson and Holling 2002).

3.3 Scenario composition

When planning for a supply of ecosystem services, any characterization of the future is subject to uncertainties in the behaviour of the actors and the unknowns of the system dynamics (Peterson et al. 2003, Cumming et al. 2005). To address this issue, scenario planning has emerged as a technique to examine the uncertainties and resilience of resource systems (Peterson et al. 2003, MA 2005, Carpenter et al. 2006, Mahmoud et al. 2009). Scenario planning was originally developed for strategic planning and war games after the Second World War (Kahn and Wiener 1967). It is now used extensively for business decision making, assessing the impacts of climate change, and as the basis for environmental risk assessment (Ogilvy and Schwartz 2004, IPCC 2000, EEA 2009).

The application of scenarios for the investigation of social-ecological systems integrates across environmental, economic and social dimensions, where a scenario describes a possible situation as "a structured account of a possible future" (Peterson et al. 2003). Predictions and forecasts are used in optimal decision making, so some benefit is maximized according to an expected probability distribution. However, because of the complex make up, dynamics and the extent of uncertainty associated with ecosystems, approaches attempting to optimize resource supply are considered inappropriate (Peterson et al. 2003, MA 2005). Further, ecological predictions that include the role of humans become confounded by people changing their behaviour when presented with new information (Morgan et al. 2007).

Scenarios are grounded in the past and based on a logical progression of events. The anticipated strength and direction of future social and ecological forces, including the consideration of uncertainty, serve as the basis for delineating a range of scenarios for consideration. A set of scenarios is structured specifically to: lend insight into system drivers; to explore uncertainties of the system behaviour; and to identify the repercussions of current resource management decisions and knowledge gaps. Scenarios are not designed to support one specific future, but instead assist in the development of management policies, that will increase the chance of achieving a socially desirable future condition (Peterson et al. 2003).

The social-ecological system's current and future forces and cycles of change provide the basis for constructing the scenarios. Because the future is uncertain, an infinite number of scenarios could be constructed (Carpenter et al. 2006). However, having a limited number of scenarios has the advantage of being easy to understand and communicate (Peterson et al. 2003, Ogilvy and Schwartz 2004, MA 2005).

Some system trends may show up in all of the scenarios, while others may be specific to one particular scenario. A "systems perspective" is used to deepen the scenario description by identifying interacting forces and trends that form a consistent pattern of events. A narrative for each scenario provides a story line with a beginning, middle and end, and can be populated with illustrative characters to personalize the plots (Olgivy and Schwartz 2004). For convenience, scenarios are given descriptive names, have a unique identity that is the result of a particular pattern of events, and reflect specific ecological forces and human management decisions.

4. Social-ecological system description and scenario composition example

I present an example of the general framework designed to implement a social-ecological approach to resource management. The case study illustrates the broader social-ecological concepts and how they can be practically applied, including the components and relationships of the social-ecological system, fast and slow drivers of change, and feedbacks. The example concludes with the composition of a set of future scenarios, based on the study area's social and ecological variability and uncertainty, designed to reflect the behaviour of the social-ecological system and the mechanisms of change.

The study area is located in southeastern BC, and encompasses 1.24 million hectares within the Cranbrook Timber Supply Area (TSA) (Figure 2-2). The area has been experiencing an unprecedented MPB outbreak (Safranyik and Wilson 2006) and provides a good example of a social-ecological system in transition from a historic configuration to some future arrangement. There are concerns about the continued supply of ecosystem services, primarily old forest ecosystems (coarse-filter biodiversity), timber for harvest and grizzly bear natal areas (fine-filter biodiversity). Old forest is being lost to MPB outbreaks, impacting the area's supply of timber and old-growth forests.

Figure 2-2. Location of the Cranbrook Timber Supply Area (TSA) in southeastern BC, Canada (Robinson 2004).

4.1 Current Conditions

The Cranbrook TSA is dominated by the Rocky Mountain Trench, with steep mountains on either side. The area is ecologically varied, but the forest is dominated by lodgepole pine; it also has a regionally significant grizzly bear population (Robinson 2004). Over the last century the disturbance regime across the Cranbrook TSA has shifted from fire to being dominated by forest harvesting. This transition has been partially supported by an effective fire suppression program (Daniels et al. 2007). An extensive road network has developed in the area to support forest harvesting. The economy is dominated by the public sector, tourism, mining, agriculture and forestry. Timber harvesting is extensive with an annual cut of 941,000 cubic meters (Robinson 2004). Beginning in 1976, a MPB outbreak emerged in the eastern part of the TSA. This outbreak peaked in

1980 and subsided by 1984. An area of approximately 150,000 hectares was affected (Young 1988). MPB became epidemic again in the late 1990s and continues today.

Direction for forest management in the Cranbrook study area is provided by a set of government and forest company management plans. These set out the various rules and regulations that are followed to meet a range of industrial, recreation and conservation interests. The economic focus is on timber for harvest. The social interest is roads and their implication for back country access and, by association, grizzly bear/human conflict. Ecologically, forest structure, pattern and composition are the main concern. The key relationships are between landscape dynamics (MPB and fire), forest management (fire suppression, roads and harvesting), wildlife habitat and old forest.

There has been an increase in timber harvesting and salvage activity in response to the MPB, and the number of roads and amount of traffic has increased. Consequently, this has led to an increase in negative grizzly bear encounters with humans (Nielson et al. 2004). As bear mortality is positively correlated with human encounters (Herrero 1985, Mattson 1990, Nielsen et al. 2004), wildlife managers have advocated that a beneficial strategy for grizzly bear conservation is to have large areas that are "secure" from human encounters (Interagency Grizzly Bear Committee 1998). These security zones are defined as areas that have adequate habitat with a minimum of human use. Their minimum size is considered to be the amount of area necessary to meet daily average foraging requirements of a female adult bear (Gibeau et al. 2001).

4.2 Future Conditions

The future conditions of the Cranbrook study area are shaped by the drivers of change, the speed with which they act, and their feedbacks. To understand how the system may change in the future, I identified the social and ecological forces acting on the region. As well, I used the concepts of the adaptive cycle and the potential for state transitions as a basis to place the forces of change in a larger system dynamics context.

Forces of Change

The Cranbrook TSA has shifted from a historic regime dominated by fire to one controlled by forest management activities and an increase in MPB activity. However, future MPB outbreaks will likely be limited by a declining availability of old pine. Fire, however, could become more prominent on the landscape with climate change. This will further reduce the amount of old pine, and thus lower the risk of future MPB outbreaks. In an effort to maintain the historic flow of timber, forest managers could respond to these disturbance events aggressively through vigorous salvage operations and massive investment in stand tending to encourage re-establishment of impacted stands (Millar et al. 2007). Alternatively, the role of forestry may be eclipsed by other values that are more conservation oriented. A more passive approach to forest management could lead to increases in old pine and risk of further MPB outbreaks.

The slow ecological drivers of interest in the Cranbrook study area are forest growth (particularly aging pine) and a slowly changing climate, which increases likelihood of drought, warmer winters, and area burned by wild fire. Fast ecological drivers include MPB outbreaks and fire. Human drivers of the system include forest management activities: rate of harvest and salvage policies (as slow drivers), and road construction, a fast driver.

There are a number of drivers that have positive or negative feedbacks in the Cranbrook. For example, MPB is a negative feedback: as MPB kills old pine the chances for further attack become diminished. Forest harvesting is a positive feedback: once harvesting activity becomes established, processing facilities and employees develop an expectation of a continued supply of timber. Similarly, there would be inertia to any change in fire suppression policy because of the risk to the standing crop of trees and infrastructure. Human visitation is also amplifying: once an area is developed there is incentive to use the roads for stand tending, salvage, recreation, etc. Now that the low-elevation areas of the Cranbrook TSA are populated, a positive feedback reinforces the increased need for land that supports recreation, tourism, grazing and agriculture. Hence, future change may include the harvest of non-timber forest products, eco-tourism, and increased value of standing trees for carbon storage. Table 2-2 lists the main forces for the Cranbrook study area, outlining the key system drivers and stressors that are relevant to timber supply, and to coarse- and fine-filter biodiversity.

Table 2-2. Key ecological and social drivers and stressors in the Cranbrook study area that can direct alternative social-ecological states of the system.

Drivers	Speed	Feedback	Description
Forest growth and succession	Slow	Positive	Continuous forest growth and succession
Timber harvesting	Fast	Positive	Forest harvesting rate and extent set by forest managers
Fire suppression	Slow	Positive	Dampening of areal extent of fire
Roads	Fast	Positive	Road building and use associated with industrial and recreational activity
Fire	Fast	Positive	Landscape-scale fire that affects long-term landscape composition and structure
MPB outbreak	Fast	Negative	Current 25-year outbreak killing of mature pine and longer term MPB dynamics
MPB sanitation harvesting	Slow	Negative	Focusing harvest on pine stands susceptible to MPB
Salvage harvesting	Fast	Negative	Level of aggression of salvage harvesting
Maintenance of landscape scale biodiversity	Slow	Positive if enhanced, Negative if relaxed	Enhancing or relaxing of landscape scale biodiversity objectives, such as increasing or decreasing harvest rotation length or additional non-harvest areas
Landscape access	Slow	Positive	Limiting access to parts of the landscape to protect wildlife from negative human encounters
Climate variability	Slow	Oscillates	Climate-driven oscillations in disturbance frequency and extent
Climate change	Slow	Positive	Shifts in rate and extent of disturbance

Cycles of Change

The adaptive cycle assumes that a system will continually renew itself and re-express past cycle phases. However, it is possible that the Cranbrook study area could transform, through a large shock or by slowly changing system drivers, into a new state that is maintained by a different set of drivers. Some transition has been observed ecologically over the past century when large fires triggered an expansion of grassland. This new grassland state is maintained by reinforcing processes. Despite large fires being more recently suppressed, grasslands have persisted through fire and cattle grazing (Lefebvre 1995, Daniels et al. 2007). Under climate change it is possible that more of the Cranbrook TSA could convert to grassland as its climate becomes less favourable to forest (Hamman and Wang 2005).

Fire suppression has been effective over the past 50 years and has altered the Cranbrook TSA's landscape dynamics. Salvage logging and MPB sanitation harvesting have been the focus of harvesting activities over large areas in the Cranbrook study area (Tembec 2005) Continuing these practices into the future will likely dampen the impact of MPB. The Rocky Mountain Trench has become more human dominated, with settlements, roads, golf courses, etc. This highly managed area is now in a different state that is maintained by intensive human management.

I summarized the dynamic behaviour of the Cranbrook TSA in a systems model (Figure 2-3). This model provides a visual summary of the interactions between the system elements and the forces that may influence different states of the forest. The system model shows two characteristic states: on the left is the fire-dominated state, defined by a negative exponential forest age structure, the expected age structure of natural forests (Van Wagner 1978), and on the right is the more MPB- and management-dominated state with an age structure that tends towards a uniform distribution (Figure 2-3) (Fall et al. 2004, Puettmann et al. 2009). Forest management mediates between these two conditions. As shown in the diagram, both fire and MPB decrease forest age, and forest management increases pine susceptible to MPB through fire suppression. Salvage harvesting links MPB and fire to forest management. The large positive and negative signs indicate feedback loops, whereas the small signs show the smaller scale relationships of the system. Roads are included due to their importance to grizzly bears. The system model identifies the key drivers, defines alternative states of the system, and helps to develop contrasting scenarios. Further, the model helps guide the development of more detailed simulation models, by focusing on the important elements of the system, feedbacks and drivers of change.

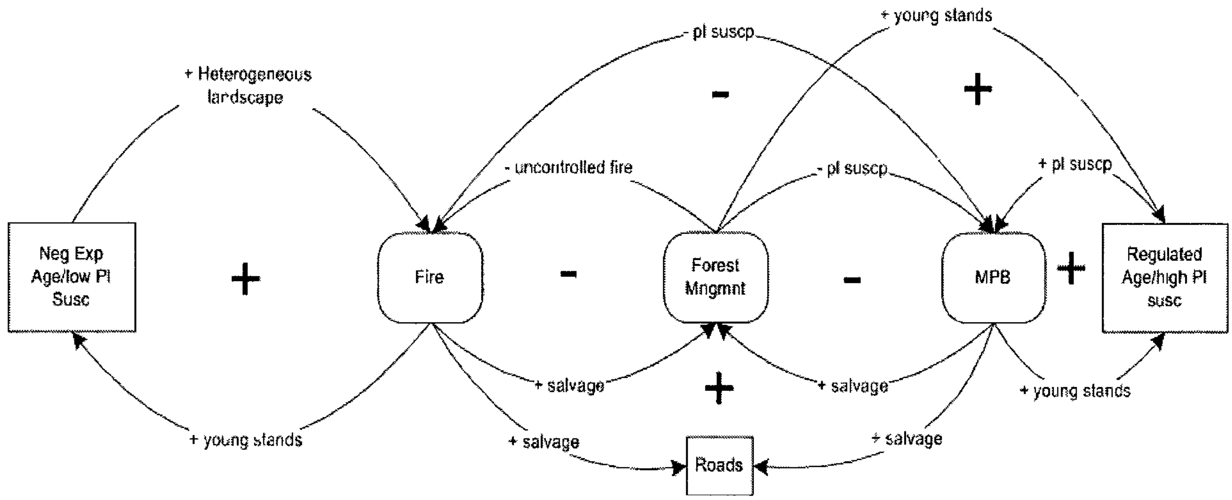

Figure 2-3. Cranbrook study area systems model showing two alternative states of the system. On the left is a natural stand age structure (negative exponential) generated by historic landscape dynamics. The right tends to a normalized age structure (uniform age structure) that results from forest management. Forest management mediates the relationship between these two opposing states. The large + and – symbols indicate positive and negative feedbacks of the various states, and the smaller signs indicate if the process in the arrow is increasing or decreasing: – implies uncontrolled fire is due to an increase in fire suppression/control efforts, while + denotes the burning of young stands converting random aged stands to young stands, creating a more negative exponential age structure. MPB converts older stands to young stands but also generates a more regulated age structure.

4.3 Scenario composition

The Cranbrook study area's ecosystem services are threatened by the current MPB outbreak and future large-scale natural disturbance events. The main social and ecological forces are identified for scenarios, with each placed on a separate axis resulting in a 2 by 2 matrix. Four scenarios are generated and they are represented as quadrants in the matrix (Figure 2-4). The scenarios capture the different possible trajectories the Cranbrook TSA could take based on the system's historical and current conditions, the fast and slow drivers of the system, and the positive and negative feedbacks. The four resulting scenarios are qualitatively different and internally consistent. The combination of social and ecological forces creates the rationale for and characterizes each scenario.

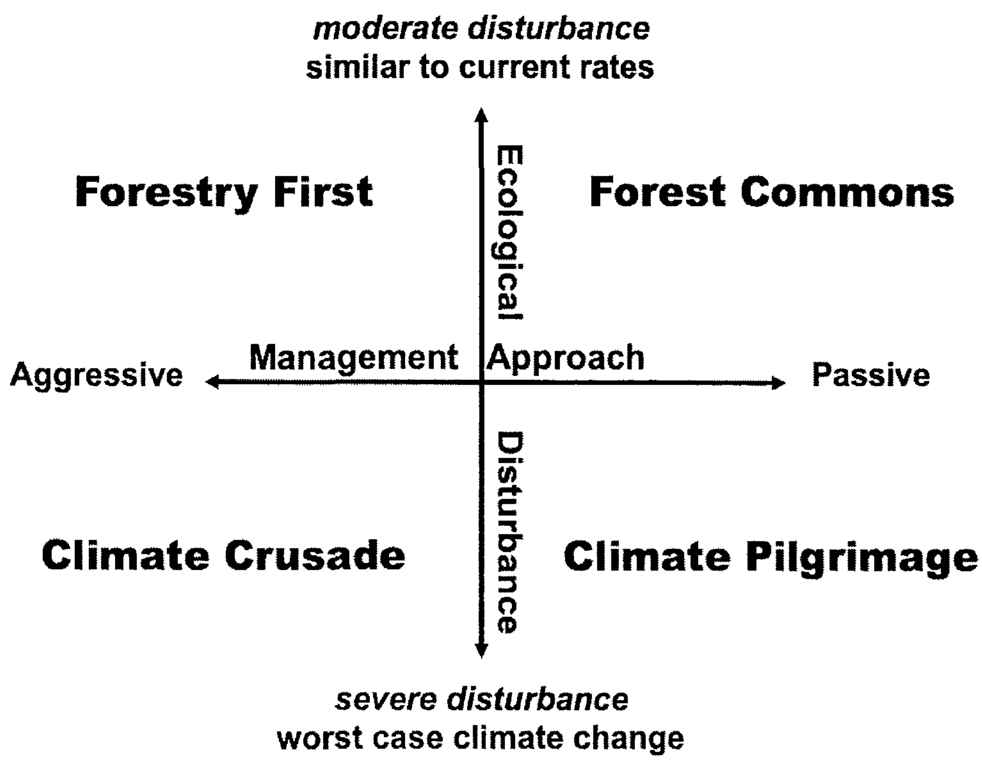

Figure 2-4. Cranbrook case study scenario matrix. The main social driver is management approach ranging from aggressive to passive. The ecological axis is defined by disturbance rates similar to current rates and severe climate change driven disturbance. The four quadrants define the scenarios: Forestry First -- aggressive harvest with moderate disturbance; Climate Crusade -- aggressive harvest with severe disturbance; Forest Commons -- passive forest management with moderate disturbance; and Climate Pilgrimage -- passive forest management with extreme disturbance.

Due to the uncertainty associated with how disturbance regimes may shift under climate change, the effect of climate on disturbance processes was identified as the main ecological axis. The ecological axis ranged from disturbance rates similar to those currently observed up to worst case increases in disturbance resulting from climate change. Socially, forest management was determined to have the largest impact on how the system may change. As a result, the social axis was summarized as "approach to forest management" and ranges from aggressive management to a more passive approach to management.

Under the aggressive approach, forest products are put ahead of other services and environmental objectives, such as preserving old forest, and access constraints do not constrain harvesting activities. Further, the aggressive management assumes that future forest condition can be controlled and any disturbance can be managed through aggressive salvage and sanitation harvest. In contrast, passive harvest attempts to restore ecological processes by limiting access to humans, constraining the total area of forest management and not suppressing fires – more of a "letting nature take its course" strategy.

Using Social-Ecological Resilience to Improve Resource Management

Humans have always used ecosystems to meet their needs. Natural dynamics can only be subjugated so much before there is a backlash: either a slow loss of environmental integrity, or an extreme disturbance event that changes the system to a form that no longer provides the same level of ecosystem service (Scheffer et al. 2001, Gunderson and Holling 2002, Carpenter 2003, Folke et al. 2004, Walker and Meyer 2004, MA 2005, Drever et al. 2006). Finding the right balance between exploiting and maintaining natural processes is a challenge society must face, especially with the uncertainty introduced by a relatively rapidly changing climate.

Current management approaches were developed during a time when forests were predominantly in the growth and conservation phases of the adaptive cycle. With recent extensive disturbance events, such as MPB, the forests of BC are in a release and reorganization phase (Burton 2010). The challenge is to shift management practices from managing for a stable accumulation of biomass to one that more prominently considers landscape disturbance and ecosystem reorganization.

The methods and ideas I present are not intended as a complete solution, but only as a contribution to a new management and planning paradigm that better describes dynamic social-ecological systems. Through these descriptions and system exploration tools, management strategies can be drafted that are more cognizant of resilience and ecological adaptation to system dynamics. Through the application of these processes, social adaptive capacity can be fostered to help people respond and manage social and ecological change.

None of the four scenarios I developed constitute a prediction of future conditions in the Cranbrook study area. Instead, the future could be a combination of elements from each, or the system could oscillate between the different system boundaries described by the various quadrants. I believe that these scenarios bound what might occur in the future and help to inform the future implications of current decisions on ecosystem services; for the Cranbrook TSA this would include timber available for harvest, coarse-filter biodiversity and grizzly bear habitat.

For the Cranbrook study area, resilience to future MPB events is enhanced through forest management activities, in which the older pine is harvested or salvaged and future stands are harvested at a younger, less susceptible age. MPB risk would continue in parts of the landscape that are reserved from harvest, but could be partially mitigated through prescribed burning to increase structural complexity. This strategy would need to be weighed against the loss of some old forest, but could be conducted such that there remains adequate representation across different ecosystem types. Risk of severe fire would remain, and even increase, with climate change perhaps triggering a transformation in some parts of the landscape to grassland. However, an option of harvesting disturbed forests for biomass, and expansion of rangelands could also be considered. Adaptation for the Cranbrook TSA could be enhanced through pre-disturbance planning and the implementation of adaptive management. Managing access would be an important component of adaptation to ensure a viable number of grizzly bear natal areas remain.

Approaches to managing for resilience have been promoted for a number of resource systems. Walker et al. (2002) suggested an alternative social-ecological approach to the management of Australian rangelands that used the maintenance of grassland resilience as a central principle. In the Northern Highlands Lake District of Wisconsin, Peterson et al. (2003) applied an ecological assessment framework, based on the Millennium Ecosystem Assessment (MA 2005), to determine the coupled social and ecological elements. The assessment framework was used to develop alternative management scenarios that were modelled and evaluated. In South Africa's Kruger National Park an approach based on monitoring “thresholds of potential concern” was used as an adaptive management strategy. When an environmental indicator was reached, action was taken to ameliorate the cause or to adjust the indicator to a more realistic level (Parr and Anderson 2006). In these projects, resilience was not directly measured, but inferred through an identification of the system's state or its identify –

namely the key components of the system, their relationships and their persistence in space and time (Cumming et al. 2005).

In general, strategies to enhance resilience may entail foregoing extensive interventions, so that the system can go through a process of renewal in order to achieve a more stable long-term state, particularly when the investment required to maintain a system in a desired state becomes overwhelming (Millar et al. 2007). This may demand managing a varied portfolio of resource systems across adaptive cycle phases; while some are predominantly producing commodities, others are less commodity oriented providing other services, such as wildlife habitat. Finally, some systems may be in renewal and left to re-organize with the expectation that at some future time they will provide higher levels of ecosystem services to communities. This proposed approach would be similar to triad land zoning, but would operate at a scale specific to the natural disturbance processes and modify zone boundaries depending on how future events unfold.

The scenario approach described here is similar to other projects interested in exploring the resilience of social-ecological systems (Peterson et al. 2003, MA 2005, Carpenter et al. 2006). This approach to scenario planning is designed to inform policies that enhance social-ecological resilience, while other scenario techniques have altogether different purposes, such as identifying the most cost-effective or efficient conservation endpoint (Lindborg et al. 2009, Koh and Ghazoul 2010). Many different scenario projects use an axis approach to identify the major forces and uncertainties (EEA 2009). However, it is novel to separate the social and ecological forces onto two different axes to reflect the social-ecological system. In general, a criticism of the axis approach is that separating and reducing the main forces into two dimensions limits the utility of the approach to deal with surprise (EEA 2009), where introducing wild cards – high-impact ecological or social surprises -- into the scenarios helps to broaden the discussion of unanticipated change that could occur (Mahmoud et al. 2009). This criticism is not relevant in the resilience context, where the scenarios are specifically focused on bounding system dynamics and sources of ecological surprise. Resilience-based scenario approaches are most useful when they explore the logical outcomes of the policy and disturbance assumptions surrounding resource systems and the provisioning of ecosystem services (Carpenter et al. 2006).

The inclusion of both qualitative and quantitative approaches for examining the future supply and uncertainty of ecosystem services has been shown to be a balanced approach to planning for the future (MA 2005). What I have presented here provides a first step in developing structured methods for designing a more detailed quantitative analysis of system forces and uncertainties. Quantitative analysis allows for a deeper investigation of system dynamics and resilience to large-scale events. A quantitative implementation of the scenarios would entail a full description of the assumptions underpinning each scenario. Simulation models would then be constructed to capture the major forces of the system: for example, forest growth, timber harvesting, roads, fire and MPB for the case study presented.

Several challenges exist in applying a social-ecological systems approach to resource management. Many of the system elements are poorly understood and lack a sufficient level of detail to fully describe. Further, there may be unknown relationships and drivers. The Cranbrook study area has only been observed in detail over the past 60 years, and there may be many dynamics that have not been recorded or expressed either directly or in the paleoecological record. A social-ecological approach shares some of the same logistical challenges facing the implementation of adaptive management. Under both approaches there would be resource manager and interest group discomfort with the level of future uncertainty that must be considered. As well, the application of entrenched "best practices" used in conventional management is inappropriate for dynamic systems. The financial and time commitment required for long-term research and monitoring of ecosystems would be difficult for jurisdictions to accept (Simberloff 1998, Stankey et al. 2003, Lindenmayer et al. 2008).

I presented a rationale for developing a social-ecological framework to describe the dynamics of complex resource ecosystems, one that captures the social and ecological drivers of change. Realizing this approach in practice would require a paradigm shift within management institutions that acknowledges the vital importance of considering a range of possible futures. The primary objective of management would be to redefine practice directives to ensure a resilient ecosystem is established, before considering the level of ecosystem services that it can provide.

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CHAPTER 3

Scenario Analysis: Landscape Dynamics, Resilience and the Supply of Ecosystem Services

Introduction

Balancing the maintenance of biodiversity and the supply of timber for harvest has led to a considerable debate over how to manage forests (Cote and Bouthillier 2000, CCFM 2003, Gerardo et al. 2005, Papaik et al. 2008). The trade-off between biodiversity and timber becomes more pronounced when forests are disturbed by natural causes such as fire or insect outbreaks (Rodríguez et al. 2006). Forest managers are challenged with making decisions that balance social and economic expectations for ecosystem services (such as timber and wildlife habitat) with a forest's capacity to adapt and respond to future disturbance. Resilience theory is promoted as a foundation for developing forest management strategies that better suit dynamic ecosystems. The resilience of a forest is its capacity to reorganize and recover from natural and human disturbance without shifting into an alternative state that is controlled by a different set of processes (Drever et al. 2006, Campbell et al. 2009, Puettmann et al. 2009).

A forest's resilience to future disturbance is a product of its structural and compositional diversity, which in turn has been shaped by historic dynamics such as fire (Peterson 2002), insects (Ludwig et al. 1978, Taylor and Carroll 2004) and wildlife (Dublin et al. 1990, Danell et al. 2003). The concepts of ecological resilience and biodiversity are highly related. Biodiversity is defined here as the diversity of genes, species, and ecosystems across landscapes (Loreau et al. 2002). Biodiversity is an essential feature of resilience, as the capacity for a landscape to recover, reorganize and adapt to change is dependent on the diversity of species and ecological communities (Bøhn and Amundsen 2004). To be resilient, an ecosystem needs to not only maintain natural levels of biodiversity, but also it must maintain its defining ecological processes, despite environmental dynamics. The resilience of a system, such as a forest, can be overcome by slow drivers of social and ecological change or by a sudden catastrophic shock. A new system then emerges, the state of which is reinforced by a different set of processes (Carpenter et al. 2003). This new system may be socially desirable. For example, converting a tract of forest to agriculture can provide more food than was previously available. Through the efforts of management, the agricultural system is maintained. However, there can be temporal trade-offs when resource benefits in the short term undermine the long-term sustainability of the system (Walker et al. 2004, Rodriguez et al. 2006)

The directive for forest managers is to maintain a consistent supply of timber for harvest. Understanding the interplay of natural forest dynamics and forest management activities is important in determining what resources the forest system can reliably produce, now and in the future. A threshold may exist between the forest's current configuration and some alternative state. There may be questions as to the alternative state's social desirability or ecological stability that may trigger changes in current management practices. From a trade-off perspective, an alternative state may supply more of a specific service and less of another, or it may provide resources in the short term, but be compromised in supplying them in the longer term. Further, the system may be slowly losing resilience to disturbance events, which could compromise the future supply of ecosystem services. The temporal resource trade-off becomes more pertinent under the expectation that forest dynamics will be even more volatile as the climate changes (IPCC 2007, Montenegro et al. 2007, Weaver et al. 2007, Williamson et al. 2009, Pojar 2010). Indeed, it is strongly advised that forest management strategies begin to integrate the ecological effects of climate change into policy and practice (Bodin and Wiman 2007, Campbell et al. 2009).

Along with managing for forest dynamics and climate change, managers must incorporate ever evolving social expectations of what ecosystem services forests should provide (CCFM 2003, MA 2005). Not only must forests provide for timber and biodiversity, but also recreation opportunities, hydrological balance and carbon storage (MA 2005). As well, there is considerable discussion about what constitutes a sustainable approach to forest management (Holling and Meffe 1996).For instance, some people are willing to tolerate an approach to forest management that downplays future catastrophic disturbance, assuming any future event can be controlled (Farrell et al. 2000). Alternately, other people are more risk averse and assume future ecological disruptions are inevitable and uncontrollable (Holling and Meffe 1996).

To explore the complex relationship between natural forest dynamics, timber harvesting and the supply of ecosystem services, it is necessary to consider the interplay of both natural (fire and insect outbreaks) and human managed processes (timber harvesting and fire suppression). Scenario planning provides a technique to support this exploration (Peterson et al. 2003, Cumming et al. 2005, and MA 2005). Scenarios can be constructed so that they capture a range of social and ecological conditions. The scenarios can then be analysed through spatial and temporal simulation models to understand different assumptions about future ecological and management dynamics and aid in the development of appropriate management approaches (Gadow 2000). Scenario planning techniques assist in shifting management from a focus on a single specific future to a focus on a range of possible futures. An analysis of a set of scenarios aids in the identification of alternative system states, and the social and ecological mechanisms of change. System relationships and feedbacks become apparent in scenario analysis and provide a rationale for further research and development. Overall, scenarios support the development of forest management policies that help build resilience to future disturbance and improve the capacity for post-disturbance ecological and social reorganization.

In this chapter, I implement a landscape simulation model to analyse the supply of ecosystem services and the resilience of a forested ecosystem to disturbance, across a range of social and ecological conditions. I use a set of scenarios to assess how a forest system can change based on different resource management assumptions and actions, and how these in turn interplay with forest dynamics. Through the scenario simulation, I provide a framework to interpret the variability and uncertainty associated with the supply of ecosystem services and system resilience.

I develop scenarios for the Cranbrook TSA in southeastern BC, an area with a forest-dependent economy and a significant grizzly bear population (*Ursus arctos*; Proctor et al. 2002). Landscape dynamics have been extensive in this area, including large historic fires and a more recent epidemic forest insect outbreak (Daniels et al. 2007). The forest industry has been challenged with minimizing the impact of natural disturbance on the timber resource.

The scenario analysis of the Cranbrook study area assesses, through the evaluation of indicators, how the system's controlling processes change, and how resilient the forest is to future disturbance. I evaluate provisioning and regulatory ecosystem services for each scenario. I use timber supply as an indicator of the provisioning service. Regulatory ecosystem services are represented by a set of indicators that evaluate fine- and coarse-filter biodiversity, which include shifts in controlling ecological processes. The indicators used to interpret each scenario reflect the concepts of resilience, biodiversity, natural forest dynamics, social expectations and climate change.

Methods

Study Area

The study area is located in southeastern British Columbia (BC) and is currently experiencing an unprecedented mountain pine beetle (*Dendroctonus ponderosae*; MPB) epidemic that is compromising the supply of ecosystem services (Taylor and Carrol 2004, Eng et al. 2005, 2006, Safranyik and Wilson 2006). The challenge for the Cranbrook study area is to manage the forest for the supply of ecosystem services that fluctuate through time due to variations in ecological and social processes. The primary social processes acting on the forested land base are timber harvesting, road development and fire suppression. These social processes interact with climate change and influence the ecological processes of fire and MPB outbreaks.

The Cranbrook TSA is in the BC Ministry of Forests Rocky Mountain Forest District and encompasses approximately 1.48 million hectares of crown and private land (Figure 2-2) of which 69% is forested. The area is split into two forest management units: a provincially managed timber supply area making up 90% of the forested area and a non crown privately managed forest (PMF).

The Cranbrook study area is bounded by the Rocky Mountains in the east, the Purcell Mountains in the west, the Skookumchuck Valley to the north and the Canada-U.S. border to the south. The area is dominated by the Rocky Mountain Trench, which cradles the Kootenay River. The west side of the trench has low foothills that rise up to the Purcell Mountains, whereas the east side ends abruptly at the Rocky Mountains, which are rugged with glaciers and steep-sided valleys.

The study area is comprised of 8 ecosystems, classified as zones under the biogeoclimatic ecosystem classification (BEC) system (Meidinger and Pojar 1991). The Bunchgrass (BG) and Ponderosa Pine (PP) zones occur in the valley bottoms, which are dry and hot with extensive grass cover; the climax overstory tree species is ponderosa pine (*Pinus ponderosa*). The Interior Douglas-Fir (IDF) zone occurs at higher elevations, from 800 to 1200 meters; Douglas-fir (*Pseudotsuga menziesii*) dominates. Between 1200 and 1600 metres lodgepole pine (*Pinus contorta*) takes over in the Interior Cedar-Hemlock (ICH) and Montane Spruce (MS) zones, above which Engelmann spruce (*Picea englemannii*) and subalpine fir (*Abies lasiocarpa*) occur up to 2000 metres in the Englemann Spruce-Subalpine Fir (ESSF) zone. Above 2000 metres is the Alpine Tundra (AT) zone, characterized by stunted trees or krummholz, rock, ice and grassy meadows. The study area has high concentrations of ungulates such as elk (*Cervus canadensis*), mule deer (*Odocoileus hemionus*), whitetail deer (*O. virginianus*), moose (*Alces alces*), Rocky Mountain bighorn sheep (*Ovis canadensis*), mountain goat (*Oreamnos americanus*), as well as, black bear (*Ursus americanus*), grizzly bear, cougar (*Puma concolor*) and lynx (*Lynx canadensis*) (Robinson 2004). The Cranbrook grizzly bear population has been extensively studied over the past 25 years. Studies include grizzly bear habitat use (McLellan and Hovey 2001), genetic isolation of sub-populations (Proctor et al. 2002), and grizzly bear response to various aspects of forest management (Apps et al. 2004).

Historically, the Cranbrook study area was dominated by fire. However, over the past 70 years wildfire has been suppressed and the lower elevations have been extensively modified by people and livestock. There are 47,000 people living in the study area, mainly in the three cities of Cranbrook, Kimberley and Fernie. The economy is dominated by the public sector, tourism, mining, agriculture and forestry. Timber harvesting is extensive with 941,000 cubic meters currently allocated to be cut annually in the TSA (Robinson 2004). Beginning in 1976, a MPB outbreak emerged in the eastern part of the Cranbrook TSA. This outbreak peaked in 1980 and subsided by 1984. An area of approximately 150,000 hectares was affected (Young 1988). An epidemic initiated in the late 1990s continues today.

Direction for forest management in the Cranbrook study area is provided by the BC Forest and Range Practices Act, the Kootenay Boundary Higher Level Plan Order, the Kootenay Land Use Plan Implementation Strategy, and the eastern portion is directed by the Southern Rocky Mountain Management Plan. The various rules and regulations provided by these plans are reflected in the Cranbrook Timber Supply Area Timber Supply Review #3 (TSR III) Analysis Report and Tembec's PMF plan, which describes the specific management strategies to achieve the planning objectives (Tembec 2005, Robinson 2004).

Scenario Analysis

Assessment overview and modelling description

I used the Spatially Explicit Landscape Event Simulator (SELES; Fall and Fall 2001) to develop simulation models for the Cranbrook study area. This software is a flexible, transparent tool for building, processing and verifying grid-based, spatio-temporal models. I parameterized each simulation model to reflect each scenario's basic management and ecological assumptions.

Input data

The majority of data were provided by the BC government and originally used for the government's 2004 Timber Supply Review process (Robinson 2004). These were supplemented with more recent government data outlining current land use planning decisions, including spatial information describing motorized and non-motorized access, old growth management areas and conservation areas. Additional data on current road use, high-value conservation forests and site-level ecosystem mapping were provided by a local division of the Tembec forest company. The model had a spatial grain of 1 hectare, which provided an adequate level of detail to assess timber supply, wildlife habitat and ecological representation of old forest. Spatial entities below this resolution such as stream buffers and roads were modelled as a percentage of a cell. The dynamic sub-models interacted on a decadal time step. I used a Monte Carlo simulation to capture the stochasticity of the natural disturbance and harvesting regimes, generating 10 data sets over a 500-year period for each scenario.

Key state variables

A set of key state variables was used to control model behaviour and to estimate the status of ecosystem services, which included indicators of system dynamics and resilience. Some of the state variables were dynamic and updated by the process sub-models: tree age and species were modified by the fire, MPB, and timber harvesting sub-models. Volumes of timber were calculated by using lookup tables that listed the amount of volume for different forest/age/productivity state variable combinations (Robinson 2004). Timber volume was used to guide the model when determining the amounts and areas to harvest. To parameterize the fire model, a lookup table was used to calculate fire return intervals for each broad ecosystem group (see fire sub-model description below for more detail). Several map layers were used as state variables to partition the landscape into zones designating where timber harvesting could occur, conservation areas, and other land designations that excluded timber harvesting. As well, watershed boundaries were used, in some instances, to limit access by restricting the number of watersheds that could host industrial activity.

State variables, or their derivatives, were used to generate indicators of the supply of ecosystem services. In addition to controlling model behaviour, timber volume was a model output. Status of forest age and ecosystem group was used for the coarse filter analysis (see coarse-filter biodiversity in Scenario Analysis section below for more detail). Road and road type (primary, secondary and tertiary) were used to direct harvest location and calculate the availability of grizzly bear habitat. This variable was updated throughout each simulation as a product of harvesting. The age and species state variables were used to calculate areas of forest susceptible to MPB.

Sub-model descriptions

Fire

Stand-replacing fire was modelled with disturbance rates and patch sizes parameterized according to their natural disturbance regime, termed here as natural disturbance type (NDT; Province of BC 1995). The Canadian Forest Service compiled a historic data set, which recorded date, ignition sources and area burned between 1919 and 2000 (S.W. Taylor pers. comm.). These data were analysed to calculate disturbance rates and patch sizes for each NDT in the Cranbrook TSA (Appendix A). Some fires straddled NDTs. In these cases, the historic fires were allocated to NDTs based on the largest NDT/fire overlap. The fire parameters specify the overall disturbance cycle (e.g., 350 years) to apply within a NDT zone, as well as the number and size of patches disturbed. In each 10-year period, a patch is chosen randomly within each NDT zone for ignition. For each of these burn events, patches continue to be selected until the target for area burned is reached for that period. The burn spreads randomly from the start point, setting stand age within each burned patch to zero, and recording a portion of the standing volume as salvageable timber. Where fires cross an NDT boundary, the area burned is assigned to the NDT where the fire originated.

I identified two types of fire years, low and high, that alternate every 20 years (Appendix A) according to oscillations of the ocean-atmosphere Pacific Decadal Oscillation (PDO). The high/low cycle is derived from the warm/cool PDO cycle and 5-year lag in drought response to the shift to a PDO warm phase (Morgan et al. 2008a). Using the historic fire data, I calculated mean area burned and fire size parameters for high and low years for each NDT. These data suggest that small stand replacing disturbances were more common, but large infrequent disturbances were more likely during warm PDO phases.

I modelled the effects of climate change on the fire regime using Hamman and Wang's (2006) climate envelopes (A. Hamman pers. comm.). As the simulation progresses, those areas within an NDT where the climate envelope has shifted are assigned a modified rate of disturbance, the other areas stay constant. The model also captures the effects of climate change through a scaling factor applied to the area burned across all NDT zones. The overall increase equals 100%, which doubles the area burned under climate change compared to historic rates. The envelope shift only covers a portion of the 100% increase. To make up this shortfall the scale factor is applied, such that over the landscape there is an overall 100% increase. The increase in area burned is based on fire and climate change studies done in an adjacent area (Nitschke and Innes 2008) and elsewhere in western Canada (Flannigan et al. 2005).

Mountain Pine Beetle

Parameters for the MPB model were based on historic MPB mean outbreak size and patch size (S. W. Taylor pers. comm.). The annual outbreak and outbreak patch size followed a negative exponential frequency distribution; however, the shape of the distribution reflected a few large episodic events and smaller more typical events (Appendix B). An occurrence of MPB was dependent on the availability of host (mature pine) trees and favourable climate and dispersal conditions. Two types of MPB incidence were modelled, endemic and epidemic. Endemic outbreaks occur regularly at low levels, whereas epidemic outbreaks are substantially larger in their areal impact. To capture this effect, the data were split at the 90th percentile, following a similar methodology used by Morgan et al. (2008a) for differentiating large regional fires. Epidemic outbreaks were assigned the mean from the data above the 90th percentile, while the endemic were calculated from the remainder of the data (Appendix B).

The model uses an index of susceptibility to guide when and where an outbreak may occur. The susceptibility of an individual forest stand was estimated based on the percentage of pine, the age, the latitude and longitude, and climatic suitability (Shore and Safranyik 1992). The index is an indicator of the expected level of damage if a stand were to be attacked (Shore and Safranyik 1992). Based on the historic MPB data, small endemic outbreaks started only after the area of susceptible pine exceeded 50,000 hectares across the study area and became common after 100,000 hectares. Epidemic outbreaks were initiated when the area of susceptible pine exceeded 100,000 hectares and became common after 140,000 hectares. The model uses the amount of susceptible pine to determine a threshold of epidemic and endemic MPB outbreaks. The expression for calculating the amount of susceptible pine (SP) is:

$$SP = \sum(S \times A)$$

where

S is the susceptibility index (0 to 1.0)

A is the mapped area of a susceptibility index

The model first selects a probability from a cumulative density function, such that if the amount of pine is above a threshold it may not trigger an outbreak (to reflect the non-determinism of outbreak initiations); alternatively, an outbreak can occur even for relatively low levels of susceptible pine. However, as the size of the susceptible area increases, so does the probability of an outbreak, such that if there are successive years with no outbreak the level of susceptibility increases. Once an outbreak is triggered, the model selects the area affected from a negative exponential distribution with a mean from the historic endemic or epidemic occurrence. Following an outbreak, 100% of the cell is disturbed and set to age zero, and standing volume is flagged as timber available for salvage.

The model does not consider the density of MPB or proximity of infested trees (Shore and Safranyik, 1992). The model is intended to capture the outcome of the MPB, not the detailed process of a population build-up and dispersal. However, the negative exponential distribution of area infected by MPB does reflect the dynamics of the beetle population. As well, the model conducts a neighbourhood analysis identifying areas with high concentrations of susceptible pine. These areas are assigned a higher probability of outbreak.

The impact of climate change on MPB is represented by the susceptibility index's climate suitability layer. The model uses a default layer reflecting current climate conditions. Under climate change this layer is modified based on shifts in climate suitable to MPB (A. L. Carroll pers. com.). Estimates of future climate were estimated from global circulation models (Carroll et al. 2006).

Forest Management

The forest management sub-model was implemented using the Spatial Timber Supply Model (STSM; Fall 2002). STSM captures the same management regime, assumptions and data as used for the base case Cranbrook Timber Supply analysis completed for TSR III (BC Ministry of Forests 1997; Robinson 2004). The model was parameterized to harvest eligible stands according to a scenario-specific forest management policy. The model harvests the Timber Harvesting Land Base (THLB), which is the area of the landscape that is accessible, productive for forestry operations and not excluded from harvest due to other interests (e.g., conservation objectives such as riparian protection, wildlife tree patches, parks, environmentally sensitive areas and steep terrain, etc.). Each stand has age, species composition and site productivity information that informs volume yield curves. The harvest rate (m^3/year) is set as a central component of the management policy and dictates how much of the THLB is harvested annually. However, the actual area harvested depends on volume/age distribution in eligible stands.

The model allows for the prioritisation of specific stands for harvest or avoidance. This includes stands for salvage, those that are highly susceptible to MPB, or ones to be avoided such as areas adjacent to conservation zones. The harvest model has a salvage component that recovers a portion of the disturbed timber based on disturbance history and shelf life -- the length of time timber volume remains merchantable following disturbance. The salvage of forest stands killed by MPB is based on the estimates of shelf life provided by the Provincial MPB modelling project (Eng et al. 2006). Considering that other types of tree mortality influence these stands, including windthrow, root rot, other bark beetles, an additional volume reduction is used to capture these “non-recoverable losses” (NRL) from the THLB (Table 3-1).

Table 3-1. Volume of timber (m^3/year) in the timber harvesting land base assumed lost to disturbance agents that were not explicitly modelled as mortality from mountain pine beetle and fire for the Timber Supply Area (TSA) and the private managed forest (PMF) in the Cranbrook study area.

Disturbance Agent	**TSA (m^3/Year)**	**PMF (m^3/Year)**
Douglas-fir bark beetle	251	49
Balsam bark beetle	337	66
Blowdown / snowpress (mature stands)	11,748	2,315
Non-catastrophic in-block blowdown/snowpress	4,048	798
Non-catastrophic blowdown/snowpress (cutblock edges)	3,863	761
Non-catastrophic blowdown/snowpress (right of way edges)	5,569	1,097
Oil, gas, and mineral exploration	50	10
Coal mine spoil failures	50	10
Red-belt damage	350	69
Total	**26,266**	**5,175**

The harvesting model also generates roads. As logging progresses, one of the priorities for identifying the next stand to harvest is distance to an existing road. When a stand is harvested, the first hectare is considered a landing and a straight-line spur road connects it to the nearest logging road. The resulting road network evolves according to these simple rules (Table 3-2).

Table 3-2. Major phases, steps, and rules for the harvest sub-model. The harvest model follows a structured process when determining blocks to harvest. There are four phases to this process: stand eligibility, stand preference, harvest effects, and block spreading. Within each phase there is a set of sequential steps with specific rules identifying eligible patches for harvest.

Phase	Step	Rule
Stand Eligibility	1	limit harvesting disturbance to eligible land
	2	in the timber harvesting landbase
	3	eligible zones (age class structure allows harvesting; status updated with each disturbance)
	4	areas within 2 km of an existing road
	5	stands older than minimum harvest age
	6	stands without adjacency constraints if applied (i.e., stands not next to recently harvested stands)
	7	stands within the current priority (e.g., salvage) or partition definition
Stand Preference and Selection	8	assign priority of new harvesting to each map cell based on stand age
	9	priority or partition focus
	10	select new cell location (first map cell to harvest) based on eligibility and priority
	11	build a road from the cell to the nearest road cell
Harvest Effects	12	harvest the cell and set stand age to zero
	13	update tracking variables (e.g., annual volume harvested and seral distribution for applicable zones)
	14	reduce the area of THLB in the cell to account for new access roads and for within-block development
Block Spreading	15	spread to eligible neighbours
	16	spread until harvest and/or block size target met

Fire suppression is assumed to be effective for the study area from 1945 to present (Daniels et al. 2007). The area burned may not have been as extensive in the first half of the century as in the second had there been effective fire suppression. For the model I assumed that fire suppression would reduce area burned by 50% (DeWilde and Chapin 2006). The reduction in area burned was not allocated evenly across years. Instead, I assumed that for some years there would be no success in suppressing fires due to other events like funding availability for suppression, accessibility of fires for fire fighting or severe weather. Further, I assumed that fire suppression would be more effective during the negative PDO phase, when there is less overall fire on the land base and therefore a higher likelihood that fires could be suppressed. Specifically, the model was parameterized so that fire suppression was totally ineffective for 25% of the time in either positive or negative PDO phases. The other 75% of the time was allocated according to PDO phase with suppression more effective during the cool phase than the warm phase, resulting in an overall effect of a 50% reduction in area burned following fire suppression.

Model implementation, verification and validation

The disturbance and forest management models were verified using experimental tests and sensitivity analysis. Models were initially tested independently then combined to ensure that there were no illogical interactions between sub-models. Model validation, in this context, is defined as the model being appropriate for the intended application (Rykiel 1996). Empirical data suitable for model verification are not available for the timeframes modelled in this project. Further, the specific conditions inherent in this study area cannot be found outside of the system (Levin 1992). As a result, this project relied on conceptual and logical validation (Rykiel 1996) such that scenarios are considered hypotheses and model outputs are a product of these hypotheses (Fall et al.

2004). Results of the scenario analyses are intended to demonstrate the linkage between initial landscape and management condition and the interaction with management rules and ecological processes. The future state of the forest projected by the models is not a prediction of future forest condition, but instead is a consequence of the social-ecological assumptions as implemented for a specific scenario (Fall et al. 2004).

For the fire model, I used 10,000 year simulation periods to calculate return intervals for each NDT. These intervals were then compared to expected values (Appendix A). For some simulations, fires did not burn to their expected extent due to geographic barriers such as mountains and lakes. An adjustment factor was included to ensure that the model results matched the expected return intervals for each NDT. The MPB model was evaluated and adjusted so that over the simulation period similar levels of susceptible pine would emerge through the interaction of fire, MPB and fire suppression. The total area impacted by MPB from the beginning of fire suppression in 1945 to 2025 (the end of the current outbreak) was 245,016 hectares or 306,315 hectares per 100 years (Appendix B). This is consistent with the provincial scale MPB model that projects the current outbreak to 2025 (Eng et al. 2006). In addition, the MPB model was run with fire but without fire suppression, and the model generated an amount of susceptible pine consistent with expected return intervals (Taylor and Carroll 2004). The pine susceptibility index identified 186,573 hectares as susceptible to MPB for the study area, with 122,381 hectares in the TSA. TSR III (Robinson 2004) reported 119,040 hectares of susceptible forest stands, a 3% difference.

In general, the approach to model evaluation was adequate for the intended purposes of this study. Logical validation depends on adequate input information. This includes having a fair representation of initial conditions and model processes, as well as suitable model parameters (Fall et al. 2004).

Scenario Description

The landscape models were parameterised in accordance with a set of management and disturbance scenarios that capture the social and ecological forces of change that have the largest potential impact and present the greatest level of uncertainty for forest dynamics and ecosystem services across the study area. The scenarios were based on a social gradient that ranges from aggressive to passive management, and an ecological gradient that ranges from natural disturbance consistent with historic rates to extreme disturbance driven by climate change. The full range of drivers and uncertainties were narrowed to a 2 by 2 matrix with a social and ecological axis resulting in a set of 4 scenarios. The scenarios were labelled: Forestry First, Forestry Commons, Climate Crusade, and Climate Pilgrimage. I defined two additional scenarios: Status Quo provided a relative comparison to current management conditions and assumptions, while No Management projected the current landscape condition that did not include harvesting, fire suppression or climate change. Each scenario was designed to be plausible and internally consistent capturing key issues and uncertainties.

Forestry First

The “Forestry First” scenario has an aggressive approach to forest management and assumes that future natural disturbance will be similar to historic rates. The scenario is interventionist, where the rate of timber harvest is maximized at all costs. This system is managed under an assumption of being able to control future landscape conditions through management interventions, including fire suppression. Natural disturbance resulting from insect outbreaks is assumed to be manageable through aggressive salvage and sanitation harvesting (harvesting trees thought to be susceptible to disturbance). In response to disturbance, forest professionals take the position that a minimum of timber should be lost to disturbance and conservation constraints can be weakened, including biodiversity and landscape access objectives.

Forest Commons

The second scenario, "Forest Commons", focuses on implementation of strict conservation targets based on the historic rates and extent of natural disturbance. Forest professionals take a passive approach to management, minimize access to the forest, don't employ fire suppression and provide a smaller area for green tree harvest (live trees). However, this scenario recognizes a salvage only zone that can be harvested in response to large natural disturbance events. The goal of the Forestry Commons scenario is to reconstruct the age structure of the forest, including patch size of old forest, so that the forest is more consistent with pre-European settlement landscape dynamics. This scenario is designed re-affirm the system's historic configuration and processes, including large carnivore relationships.

Climate Crusade

Under the third scenario, "Climate Crusade", extreme climate change is present and accompanied by extensive natural disturbance and ecological shifts. This scenario is characterized by disturbance rates and extents far greater than today's, including an expansion of grasslands and the woodlands into the alpine. Under this scenario, the climate envelopes shift according to changes in temperature and precipitation forecasted in downscaled general circulation models (GCM; Hamann and Wang 2006). Massive intervention characterizes this scenario including large-scale salvage. As well, provisioning services become the focus "at all costs". This objective is facilitated by relaxing environmental objectives such as allowing harvesting in riparian areas and protected areas.

Climate Pilgrimage

Similar to the Climate Crusade scenario, "Climate Pilgrimage" also recognizes extreme climate change. Extensive access restrictions are in place to minimize human impact on changing ecosystems. The theme of this scenario is to embrace and work with ecological change. The human population across the study area declines due to a reduction in industrial and recreation activities caused by a significant proportion of the land being off-limits to any human use. In this scenario, forest professionals attempt to adapt to a changing ecological regime, including a shift away from traditional forestry to one that is focused on salvaging trees impacted by natural disturbance. Furthermore, the management of the forest for carbon storage is an explicit objective.

Indicators

I constructed indicators to track provisioning and regulatory ecosystem services under the different scenarios. The indicators I chose to evaluate the Cranbrook TSA's ecosystem services are structured to lend insights into the complex social-ecological dynamics of the scenarios and their social-ecological resilience. The long-term supply of timber for harvest represents provisioning services. Regulatory ecosystem services are embodied by fine- and coarse-filter biodiversity, which includes an assessment of controlling processes.

The presence of fine-filter biodiversity, representing critical habitat for key species (Hunter 1991), is indicated by the extent of potential grizzly bear natal areas. Most organisms and plant communities in forested ecosystems are evolutionarily adapted to the disturbance regime where they occur and have the capacity to recover after a natural disturbance event (Bergeron et al. 1999, Turner et al. 2003). Coarse-filter biodiversity (Hunter 1990) represents a broad spectrum of habitats for species, which depends on maintaining natural ecological processes and the resulting forest structure equivalent to what would occur due to natural landscape dynamics (Hunter 1990). A set of indicators is used to assess coarse-filter biodiversity, which includes the distribution of forest ages across the forest, the amount of old forest relative to what would be expected under historical conditions, the dominant disturbance process, and the amount of pine susceptible to MPB. Maintaining ecological processes and structural diversity promotes resilience to disturbance (Bergeron et al. 2002, Kuuluvainen 2002, Drever et al. 2006, Puettmann et al. 2009).

Scenario Implementation

The four scenarios have either an aggressive (Forest First and Climate Crusade) or passive (Forest Commons and Climate Pilgrimage) approach to forest management. The scenarios use different management strategies to achieve their objectives, including different harvest levels, modifications to the overall size (area) of the THLB, application of conservation targets, salvage rules and harvest priorities (Table 3-3). I present each management strategy with a description of the basic differences between the aggressive and passive approaches.

Table 3-3. Long term harvest request, Timber Harvesting Land Base and the forest management rules applied to each management scenario in the Cranbrook study area. The harvest request for Forestry Commons and Climate Pilgrimage is intentionally low with the majority of timber assumed to be from salvage.

Scenario Harvest Rule	No Mngmnt	Status Quo	Forestry First	Forestry Commons	Climate Crusade	Climate Pilgrimage
Long Term Harvest Request (m3/ha/year)	0	941,637	997,513	81,765	997,513	81,765
Timber Harvesting Land Base (ha)	0	477,040	552,853	400,867	552,853	400,867
Fire Suppression	No	Yes	Yes	No	Yes	No
Unlimited Salvage	N/A	No	Yes	Yes	Yes	Yes
Salvage Only Zone	N/A	No	No	Yes	No	Yes
Access Constraints	N/A	No	No	Yes	No	Yes
Landscape Biodiversity	N/A	Mixed Emphasis	Low Emphasis	Mixed Emphasis	Low Emphasis	Mixed Emphasis
Climate Change	No	No	No	No	Yes	Yes

Setting Harvest Level

To set a long-range sustainable yield for timber supply, for any given set of management constraints, the landscape model was run iteratively to test the effect of different annual cut rates. The iterations were structured such that they eventually converged on the maximum rate (expressed as a proportion of the current harvest level) that satisfies the long-range yield criteria of a stable amount of growing stock. As well, the harvest level declines to a level that can be consistently harvested and this decline was structured to be as consistent as possible with Ministry of Forests and Range policies (e.g., declines between decades of not more than 10%). The harvest level for the aggressive scenario is the same as the status quo. Unlike the status quo, however, the aggressive scenarios will cut above the harvest request when there is a salvage opportunity. The model will then revert back to the original harvest level, whereas harvest policy in BC would dictate establishing a lower harvest level reflecting the loss of timber from a disturbance event. The aggressive regime will always try to maximize harvest and minimize lost opportunity despite the implications for long-term sustainability. The scenario is subject to inertia and reacts to harvest opportunistically.

The harvest level for the passive scenarios differs from the aggressive ones by considering possible future disturbance and by having a smaller land base available for harvest. The harvest level is set initially to that of the status quo, but on a smaller THLB with a conventional salvage rule. The harvest level drops immediately to converge on a much reduced long-term level. This level of harvest is a minimum that a local timber processing facility can expect. Following large-scale disturbance, an unlimited salvage rule is invoked causing the harvest level to temporarily increase and expand into a salvage-only zone. The exact timing of this increase is unknown since it is an adaptive response to the disturbance event. On average, however, the overall harvest level is a combination of the minimum base level and the average amount of salvage that results from harvesting uplift in some, but not all, periods.

THLB

For the aggressive scenarios, I designated an area adjacent to the status quo THLB that could be harvested to make up volume shortfalls. This area was created by identifying portions of the study area originally netted out of the THLB due to ecological reasons, including environmentally sensitive areas prone to avalanche, high water values and regeneration issues, Old Growth Management Areas (OGMAs), riparian ecosystems, wildlife tree patches, parks, and unstable terrain. These additions add 75,813 hectares to the THLB, a 16% increase.

In contrast, the passive scenario recognizes additional area in the THLB as ecologically sensitive: patches that are adjacent to ecologically sensitive forest, as identified in the status quo THLB. Harvest only occurs in these newly designated sensitive areas if timber cannot be found elsewhere. Further, the passive scenario excludes areas considered to be of high conservation value (Stuart-Smith and Wells 2006) from green tree harvest, but allows entry for salvage. In total, the passive scenario has a green tree THLB that is 89,629 hectares smaller, a 19% decrease from the status quo (Table 3-3).

Conservation Rules

Under the Timber Supply Review, landscape level biodiversity is managed through retention guidelines for the amount of old and mature forest by landscape unit/biogeoclimatic subzone-variant combination. Each of the landscape units is assigned either a high, medium or low biodiversity emphasis option as set originally in the Province's biodiversity guidebook (Province of BC, 1995), then later in the Non-Spatial Old Growth Order (Province of BC, 2004), and adopted into local and regional land use plans (Kootenay-Boundary Higher Level Plan and Southern Rocky Mountain Sustainable Resource Management Plan). The aggressive scenarios set all landscape units to low biodiversity emphasis, thereby allowing for more harvest and less retention of old and mature forest. The passive scenarios use the same emphasis options as the status quo (Table 3-3).

As an additional conservation objective, the passive scenarios limit harvest according to operating areas. Watersheds are used as operating areas and only a certain number are allowed to be open at any one time. I assume that by limiting the spatial extent of harvest, more of the landscape will be free of frequent human use and be available as female grizzly bear natal areas. Large watersheds can represent a significant proportion of the THLB available for harvest and increase short-term harvest, but as a result decrease longer-term harvest after they are deactivated. To compensate for this area effect, large watersheds were divided into smaller units. Also, an additional constraint was added that limits the amount of THLB that can be activated within a time step. This results in the end of harvesting either when a maximum number of watersheds has been accessed, or the maximum area of the THLB has been allocated. The constraint is only applied to green tree harvest and does not affect salvage harvest.

Salvage Harvesting

The salvage sub-model calculates salvage potential based on tree shelf life and volume. Shelf life estimates are a function of the BEC zone, such that decay rates are high in wet ESSF, medium in dry ESSF and ICH, and slow in PP and IDF (Eng et al. 2006). Once the cumulative decay estimate surpasses 30% of original volume, the stand is no longer merchantable. The passive scenario uses the High Conservation Value Forests as a salvage-only zone that cannot be harvested in the absence of disturbance.

Some scenarios use an unlimited salvage mode that permits harvest above the normal level in response to disturbance. The aggressive scenarios have a slight uplift due to the unlimited salvage in the early time periods as they expand into a larger THLB, but in later time periods the scenarios are already harvesting all available timber. The passive scenarios can, in response to disturbance events, temporarily increase their harvest level and access the salvage-only zone (Table 3-3).

Scenario Assessment

The scenarios were compared using a post-simulation analysis of key response variables that track the state of each scenario and the status of the provisioning and regulatory ecosystem services. The 10 replicates for each scenario differ due to the stochastic implementation of fire and long term MPB, as well as management response to changing landscape conditions. The indicator variables are recorded for each scenario across the 10 replicates. Maps show forest stand composition, stand age, fire and logging disturbance, forest class (in terms of THLB, non-THLB, and Protected Area status), and road type (highways, secondary roads, etc.) for each decade of the simulation. The model produces a set of text files for each run that summarize volume and area harvested, road length, growing stock, various breakdowns of stand age, amount and type of forest disturbed, etc. These measures and indicators were further refined and analysed using the R language and statistical software package (R Development Core Team 2010).

Timber Supply - Provisioning Service

The landscape model output a set of timber supply indicators for post-simulation analysis. Each scenario had a specific harvest request for each 10-year period of the simulation. Due to the stochastic implementation of the disturbance model, the harvest request was not met in some periods for some simulations. Natural disturbance can erode the amount of timber available for future harvest. However, for those scenarios with unlimited salvage it was possible to exceed the harvest request during periods of extensive fire or MPB outbreak. The percentage of the requested harvest met was calculated for each period of each scenario. For each scenario, the mean and standard deviation of timber volume, area of land harvested, growing stock, and the salvage proportion of the volume harvested were calculated. The amount of volume deducted for NRLs were done proportional to the amount of THLB in a given scenario.

Fine-filter Biodiversity - Regulatory Service

Grizzly bear natal security areas were used as a fine-filter conservation indicator. Grizzly bears face an increase in mortality when they encounter humans (Herrero 1985, Mattson 1990). Thus, large areas that are "secure" from human use are beneficial for grizzly bear conservation (Interagency Grizzly Bear Committee 1998). Security areas are defined as areas that have adequate habitat with a minimum of human use. Their minimum size, 900 ha per female grizzly bear, is considered to be the amount of area to meet daily average foraging requirements (Gibeau et al. 2001). The integrity of the security area is sensitive to the extent and spatial arrangement of developments including settlements, recreation areas and busy roads. A post-simulation grizzly bear security area model used land cover type (forest, grassland, water, urban, mining, non-vegetated from provincial Baseline Thematic Mapping; Yazdani et al. 1992), elevation, and roads (based on current roads and roads modelled within the harvesting sub-model) to determine the number and size of potentially secure areas.

Roads were the dynamic information used in the analysis of natal security areas. All areas within 500 meters of a "high use" road did not contribute to security areas. Permanent roads were always high use and include paved roads and unpaved roads that connect towns or villages. Logging roads were dynamic and it was assumed that harvesting traffic will end up on one of the permanent roads en route to a mill. The model dynamically created road segments by linking harvest areas to the nearest existing road segment, which eventually flow onto a permanent road. The model estimated the number of visits on the logging roads based on harvesting activity. Activity resulting from five or more harvested hectares a month results in a road being designated high use while less than five hectares of harvesting results in a low use classification.

The natal security model first identified non-suitable habitat based on land cover (subalpine/avalanche, forest and grass) and an elevation threshold of 2500 m. The model then identified high-use roads and excludes areas within 500 metres of those roads. Patches less than 900 hectares were then excluded and the remaining patches were considered secure natal security habitat for grizzly bear. The mean and standard deviation of the total area of security patches and the number of security patches was calculated across the 10 replicates for each scenario.

Coarse-Filter Biodiversity - Regulatory Service

Coarse-filter biodiversity was represented by the distribution of forest age, amount of old forest, the dominant disturbance process, and the amount of pine susceptible to MPB. Forest age structure is a product of historic disturbance, fire suppression and timber harvesting. Shifts in age structure indicate a change in the underlying drivers of the system, from one controlled by natural processes to one heavily influenced by people. A more uniform forest distribution is a product of intensive forest management (Puettmann et al. 2009), whereas a negative exponential distribution is more commonly associated with forests subject to less human management (Van Wagner 1978). Through time, harvesting should result in the majority of the forest becoming younger than the minimum harvest age (predominately 100 years in the Cranbrook). For each scenario, I calculated the amount of area in 10-year age classes for year 250 of the simulation. I used the Kolmogorov-Smirnov test (R Development Core Team 2010) to compare the age distribution of the forest to that generated under the No Management scenario.

Old forest habitats are the most rare and therefore considered most at risk on disturbance-prone landscapes (Spies et al. 2006). For each scenario, I modeled the area of old forest in each ecosystem group (age >150 or >250 years, depending on the ecosystem) and compared it to the amount of old expected according to the historic disturbance regime. Ecosystem groups (Appendix C) were based on the work of Wells et al. (2004) and were a refinement of the provincial ecological classification system (Meidinger and Pojar 1991). Each scenario's maps of stand age, forest cover type and forest management zones (protected areas, THLB, and non-THLB) were intersected with the ecosystem map and output tables generated for post-simulation analysis. The table listed the ecosystem group, the age at which it was expected to be old, total area of old forest, and a percent threshold for amount of expected old based on the ecosystem's disturbance history.

A shift in dominant disturbance agent would indicate that the system is being controlled by a different set of processes and therefore could be considered to have changed state. The shift may be subtle with certain processes playing a larger role than previously. However, when shifts in disturbance agents are considered with other indicators, an interpretation can be made as to whether the system is functionally different. This indicator tracks the area disturbed by fire, MPB, total harvest and salvage harvest for each period of the simulation. The last coarse-filter indicator is the amount of pine susceptible to MPB, an indication of the risk that the forest may be sensitive to an outbreak. For each scenario, the mean and standard deviation of the area susceptible to MPB were calculated.

Expected Trends and Outcomes for Response Variables

The state of the provisioning and regulatory services that I identified reflects social-ecological resilience of the Cranbrook system. Considering the variation in social and ecological drivers among scenarios (Table 3-3), I predict specific trends and outcomes for each response variable. An even supply of timber and a stable growing stock implies consistency in the supply of the provisioning service. I would expect this with the Status Quo and Forestry First scenarios, whereas provisioning services for the Climate Crusade scenario will likely decline due to climate change. The Forestry Commons and Climate Pilgrimage scenarios should maintain a low level of timber supply, but respond to harvest opportunity and, on average, harvest above this level, with Forestry Commons more than Climate Pilgrimage. I would expect growing stock to be stable under the Status Quo and Forestry First scenarios, increase under the Forestry Commons and Climate Pilgrimage scenarios, and decline under the Climate Crusade scenario.

Grizzly bear natal habitat reflects the amount of the study area that is not dominated by humans. I expect the amount of natal habitat to be greater under the Forestry Commons and Climate Pilgrimage scenarios. A negative exponential forest age structure, and an amount of old forest similar to historic levels, would indicate maintenance of ecosystem processes and the resilience of the landscape. I would expect this outcome with the

Forestry Commons and Climate Pilgrimage scenarios. Resilience will also correlate with less timber susceptible to MPB. I would expect this outcome for the Status Quo, Forestry First, Climate Crusade and Climate Pilgrimage scenarios.

Results

Harvest Indicators - Provisioning Service

The percent harvest achieved (Figure 3-1) shows that the Status Quo harvest request was met for over half of the 10-year time periods (29 of 50 periods). The Status Quo harvested on average 93% of the expected area; however, it only harvested 72% of its targeted volume. The volume shortfall was due to the difference in frequency and extent of disturbance between the TSR III's NRL approach and the spatially explicit dynamic disturbance models implemented for the scenario analysis. The NRL approach for setting harvest requests reduces volume by an annual uniform amount per period based on recent disturbance data, whereas the scenario analysis used the stochastic spatial MPB and fire model parameterized with longer-term historic data.

Due to the larger THLB, unlimited salvage feature and ability to violate conservation objectives when there are harvest shortfalls, the Forestry First scenario moderately exceeded its area harvest request for 36 periods (105% of area requested harvested on average). Despite the more aggressive harvesting approach used in Forestry First, like the Status Quo scenario, less volume was harvested than targeted (83%). The response to salvage opportunities was most pronounced in the Forestry Commons scenario where harvested area exceeded the request for 42 periods. Overall, the salvage-only zone and unlimited salvage features of the Forestry Commons scenario

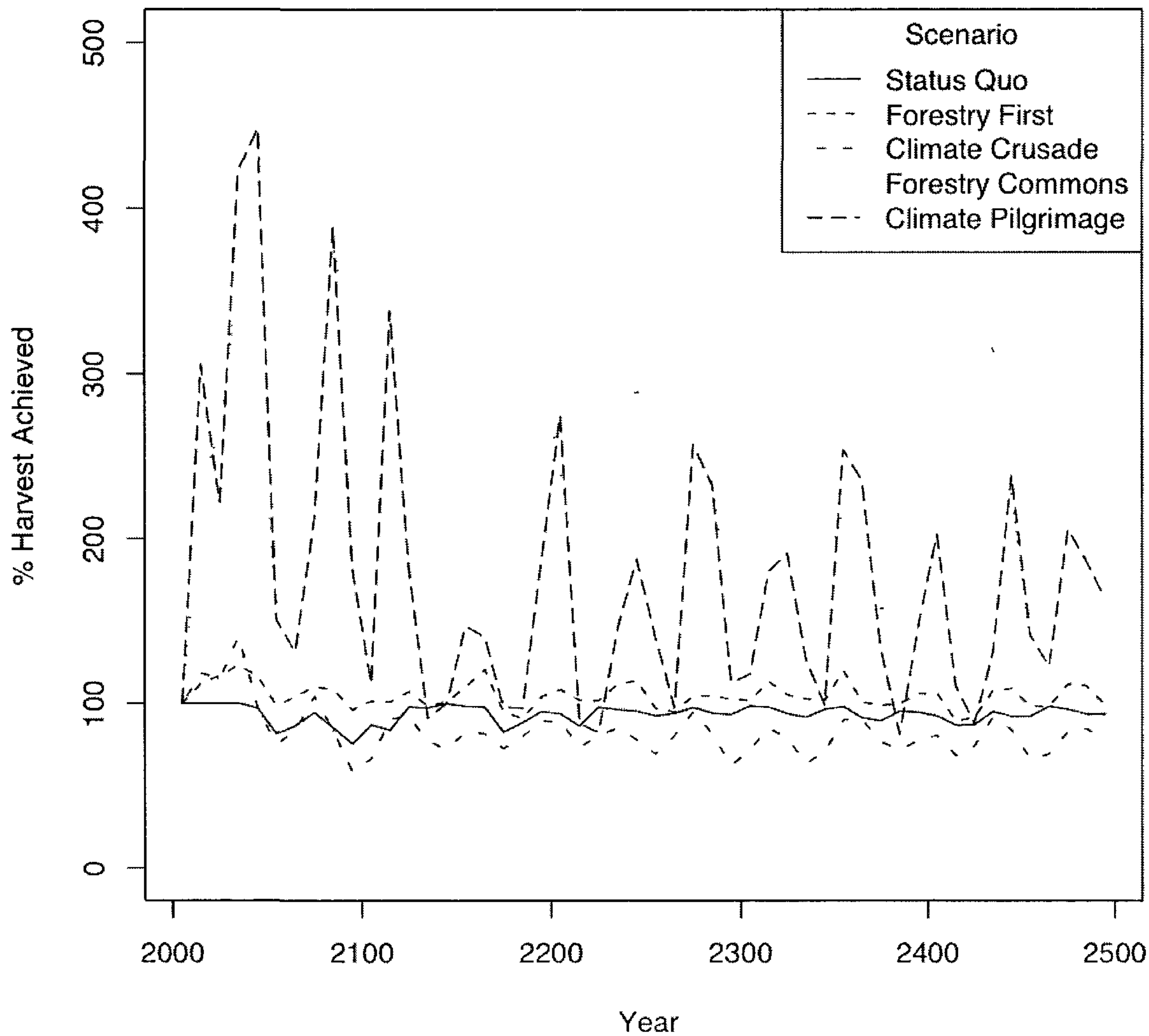

Figure 3-1. Cranbrook study area mean per cent of harvest targets achieved by year for 500 years. Simulations were repeated for 5 scenarios (Status Quo, Forestry First, Climate Crusade, Forestry Commons, and Climate Pilgrimage) representing a range of management approaches and disturbance rates.

allowed harvest of 240% of the area targeted and 232% of the volume. Following climate change, the harvest under the Climate Crusade scenario did not meet the total area (83%) or volume (58%) of requested timber, whereas under the Climate Pilgrimage scenario, harvest is 176% of the area and 140% of the volume requested.

The timber volume harvested is variable for all scenarios (Figure 3-2). The Status Quo, Forestry First and Climate Crusade scenarios all begin with the current cut level. The Status Quo roughly follows the prescribed harvest profile, but falls below due to the stochastic implementation of fire and MPB. The harvest request for the Forestry First and Climate Crusade scenarios are the same as the Status Quo, however they both increase volume harvested in response to salvage opportunity and the larger THLB, after which the Forestry First ($\bar{x}$ = 834,699 m^3, sd = 203,060) cycles somewhat above the Status Quo ($\bar{x}$ =733,079, sd = 134,392) and the Climate Crusade below ($\bar{x}$ = 585,914, sd = 259,192). The Forestry Commons ($\bar{x}$ = 384,999, sd = 263,264) and Climate Pilgrimage ($\bar{x}$ =232,403, sd=249,622) have a lower harvest request and the volume harvested increases due to salvage and then oscillates with future disturbance for the remainder of the simulation. These results presented are an average across 10 replicates of each scenario, which dampens the variability characteristic to an individual run. Under climate change, the Climate Pilgrimage scenario harvests only 60% of the Forestry Commons scenario and the Climate Crusade harvests 70% of the Forestry First scenario on average. The proportion of salvage (Figure 3-3) shows that the Forestry First has marginally more salvage than the Status Quo scenario on average (8% vs. 11%) and that under climate change the Climate Crusade salvages a greater proportion (16%). Compared to the other scenarios, both the Forestry Commons and Climate Pilgrimage employ a substantially greater proportion of salvage as part of the total volume harvested (60% and 70%).

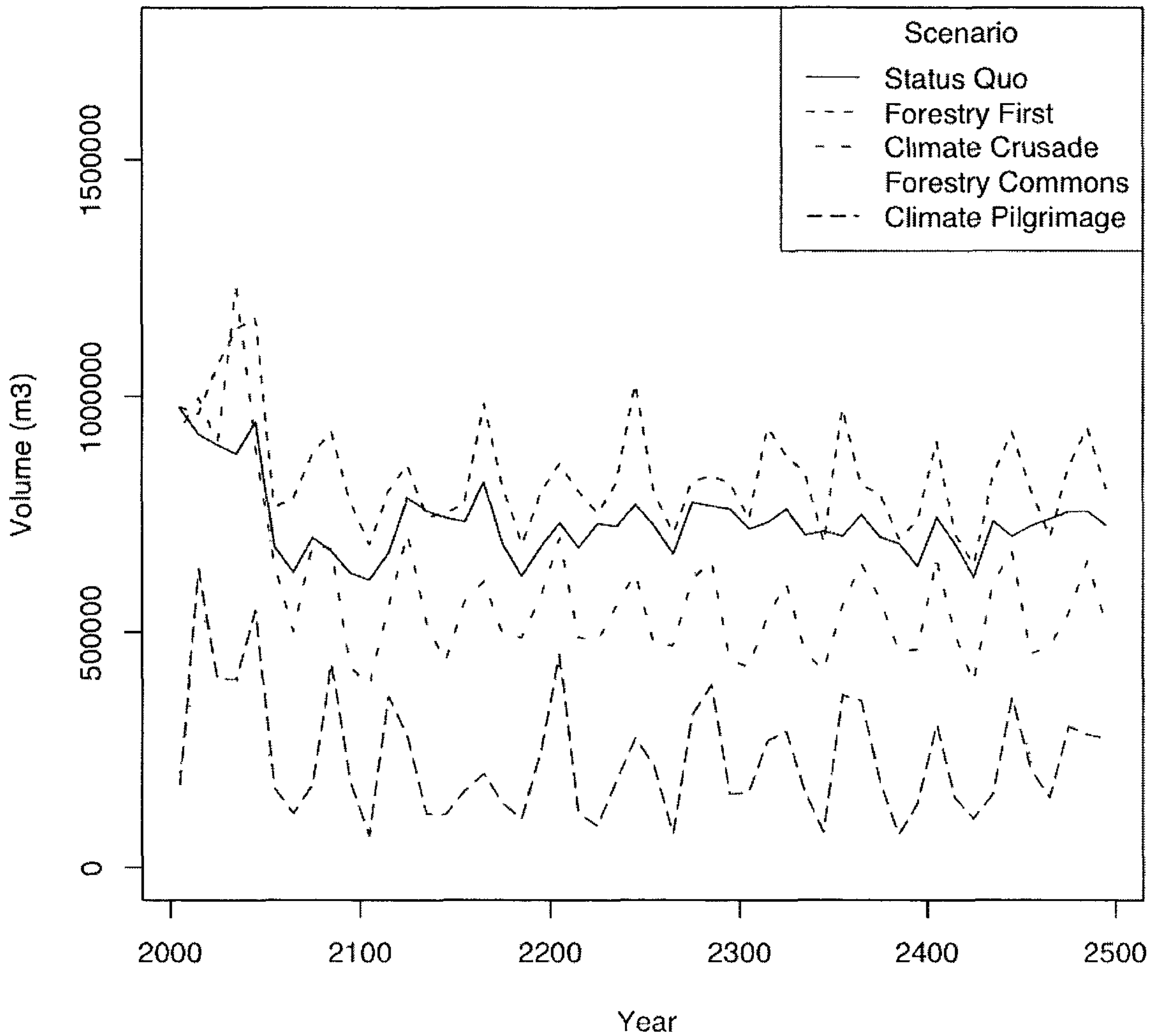

Figure 3-2. Cranbrook study area mean volume harvested over 10 replicates for 500 years of simulation. Simulations were repeated for 5 scenarios (Status Quo, Forestry First, Forest Commons, Climate Crusade and Climate Pilgrimage) representing a range of management approaches and disturbance rates.

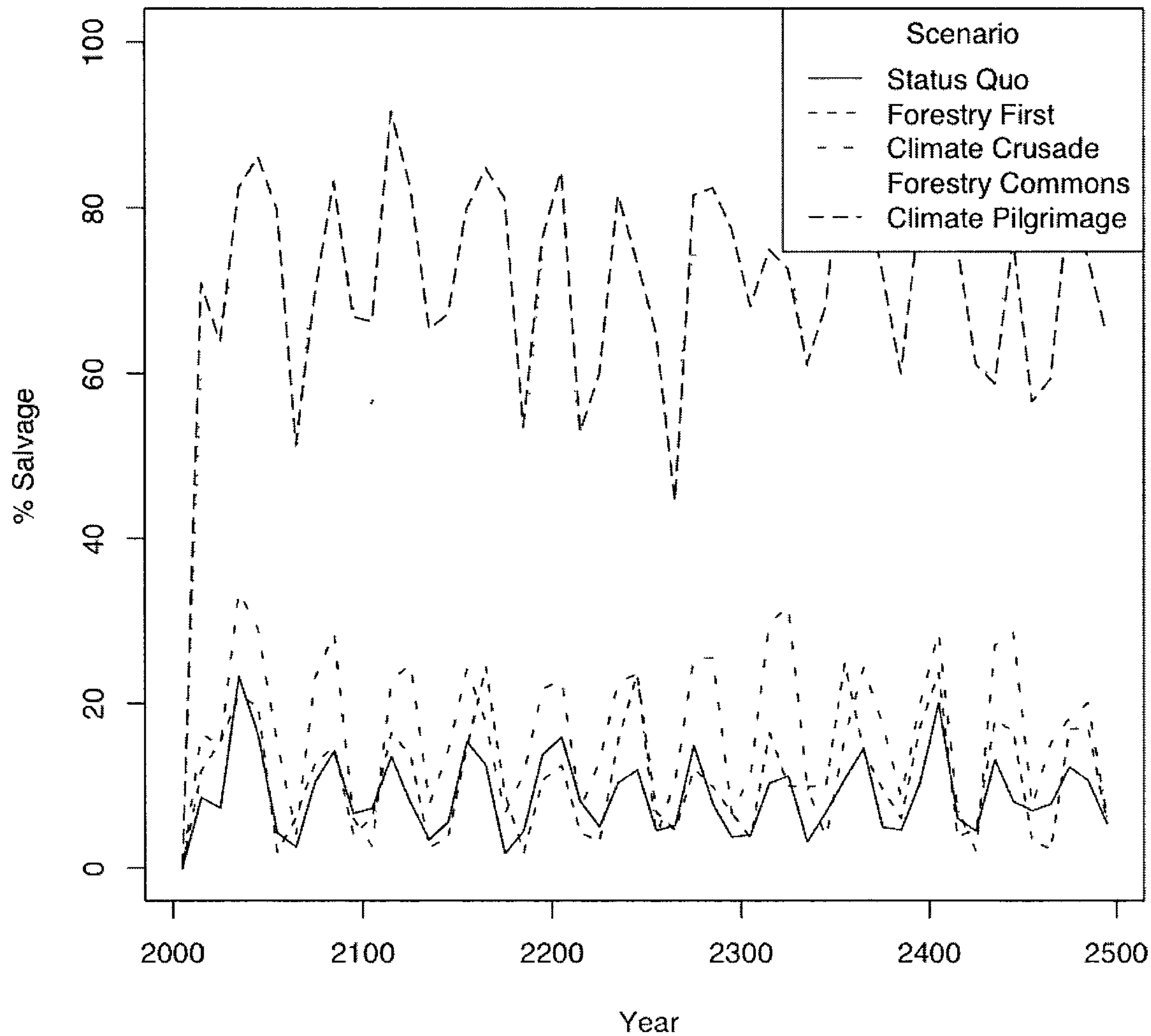

Figure 3-3. Cranbrook study area per cent salvage harvested over 10 replicates for 500 years of simulation. Simulations were repeated for 5 scenarios (Status Quo, Forestry First, Forest Commons, Climate Crusade and Climate Pilgrimage) representing a range of management approaches and disturbance rates.

Although still fluctuating, the Status Quo scenario has the most stable growing stock of the 5 scenarios ($\bar{x}$ = 37,990,786, sd = 5,516,112; Figure 3-4). Oscillations in the amount of growing stock are due to either harvesting above sustainable levels, for the Status Quo and Forestry First ($\bar{x}$ = 46,972,342, sd = 7,748,092) scenarios, or higher levels of disturbance in combination with harvesting for Forestry Commons ($\bar{x}$ = 58,001,329, sd = 10,361,347), Climate Crusade ($\bar{x}$ = 35,611,988, sd = 9,816,722) and

Climate Pilgrimage ($\bar{x}$ = 34,204,790, sd = 12,570,810). The Forestry First scenario has a higher growing stock than the Status Quo due to its larger THLB. Despite a smaller THLB, the Forestry Commons's growing stock is higher due to less green tree harvest and influence of the salvage-only zone. The two climate change scenarios both have lower growing stock than the Status Quo due to the increased incidence of disturbance killing trees that would have otherwise contributed to the growing stock. Once the oscillations begin, by over-harvesting or by natural disturbance, they persist leading to a "boom and bust" effect. The standard deviations of the growing stock of each scenario illustrate the variability across the 10 replicates. The large positive and negative iterations of growing stock are cancelled by averaging across the replicates in generating the mean. To highlight the variability in growing stock a single run is presented (Figure 3-5). The single run is the replicate with a mean growing stock closest to the overall mean across all replicates.

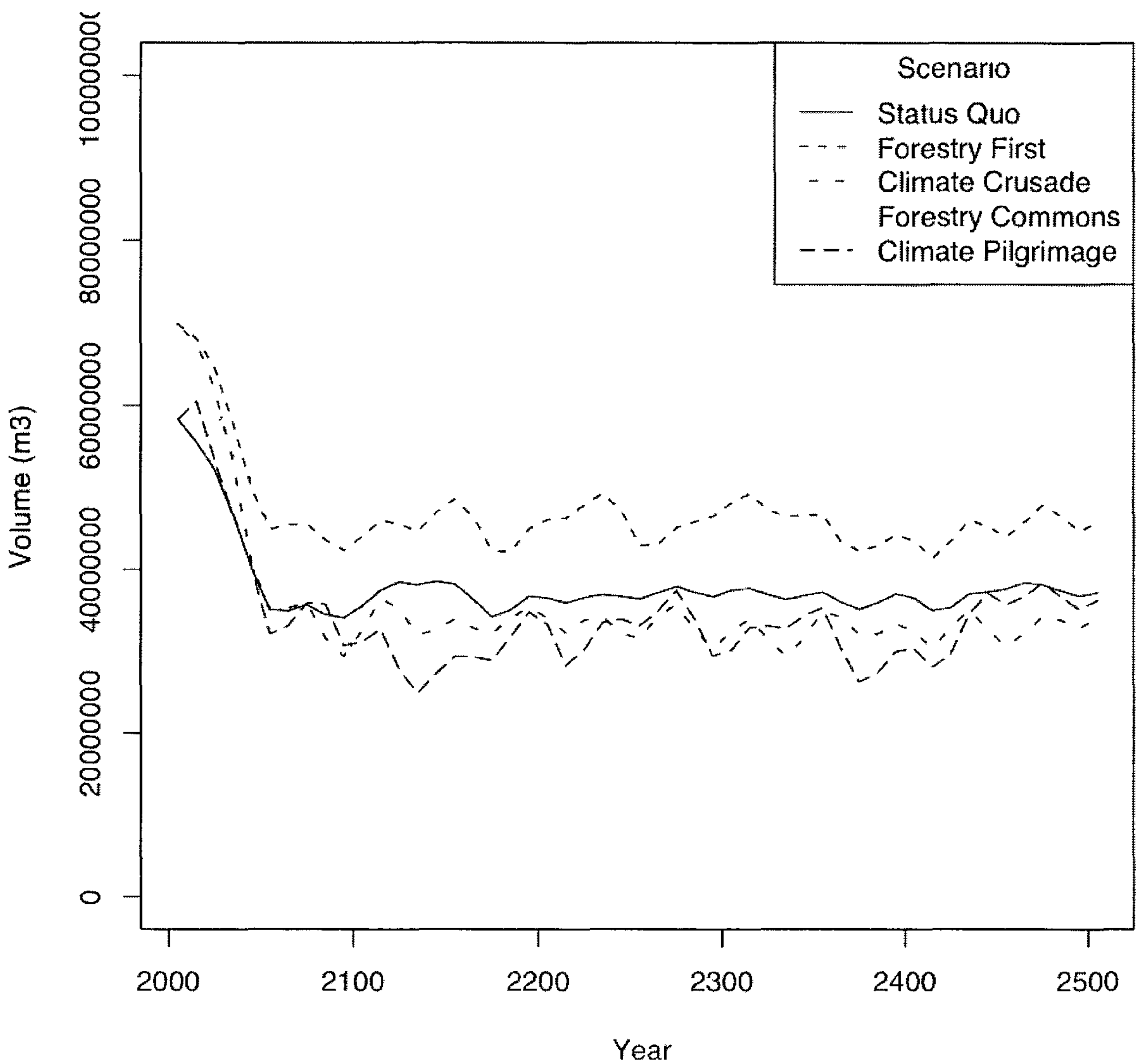

Figure 3-4. Cranbrook study area growing stock over 10 replicates for 500 years of simulation. Simulations were repeated for 5 scenarios (Status Quo, Forestry First, Forest Commons, Climate Crusade and Climate Pilgrimage) representing a range of management approaches and disturbance rates.

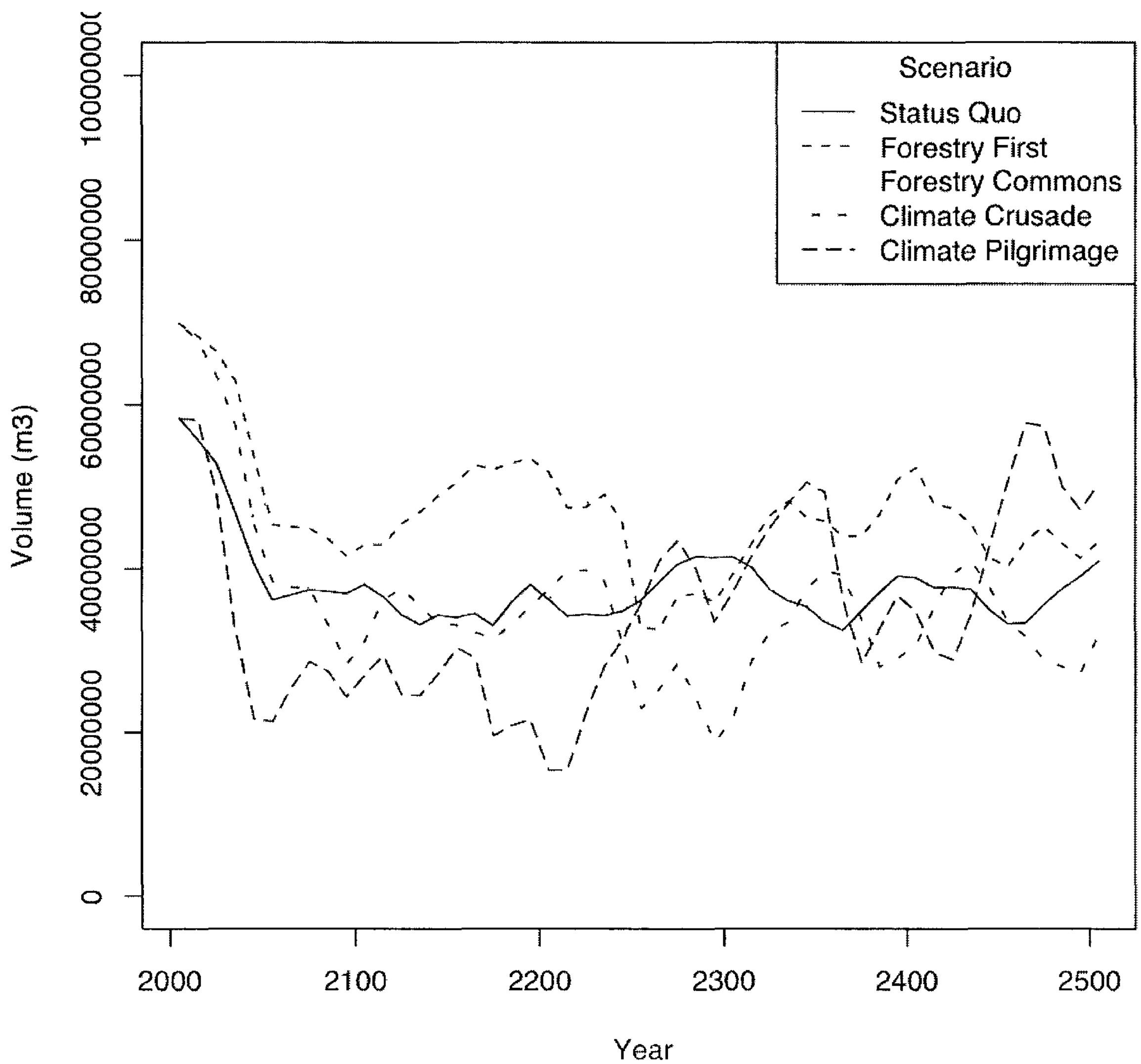

Figure 3-5. Cranbrook study area growing stock for one replicate for 500 years of simulation. A separate replicate (approximating the mean of all 10 runs) is presented for each of 5 scenarios (Status Quo, Forestry First, Forest Commons, Climate Crusade and Climate Pilgrimage) representing a range of management approaches and disturbance rates.

Fine-filter biodiversity - Regulatory Service

Prior to extensive harvesting the total area for female grizzly bears secure from human activity comprised over 80% of the study area and has declined through the 20th century (Figure 3-6). Both the Forestry Commons and Climate Pilgrimage scenarios have rules limiting access for forestry activities; these rules in conjunction with a lower harvest provide for substantially more secure areas (911,369 ha and 933,350 ha respectively) than the Status Quo, Forestry First or Climate Crusade scenarios (740,736 ha, 726,692 ha, and 772,777 ha). The Climate Crusade scenario has 6% more security area than the Forestry First scenario due to the increase in salvage harvesting, which tends to concentrate harvesting. The number of areas for female grizzly bears secure from humans shows a similar trend (Figure 3-7).

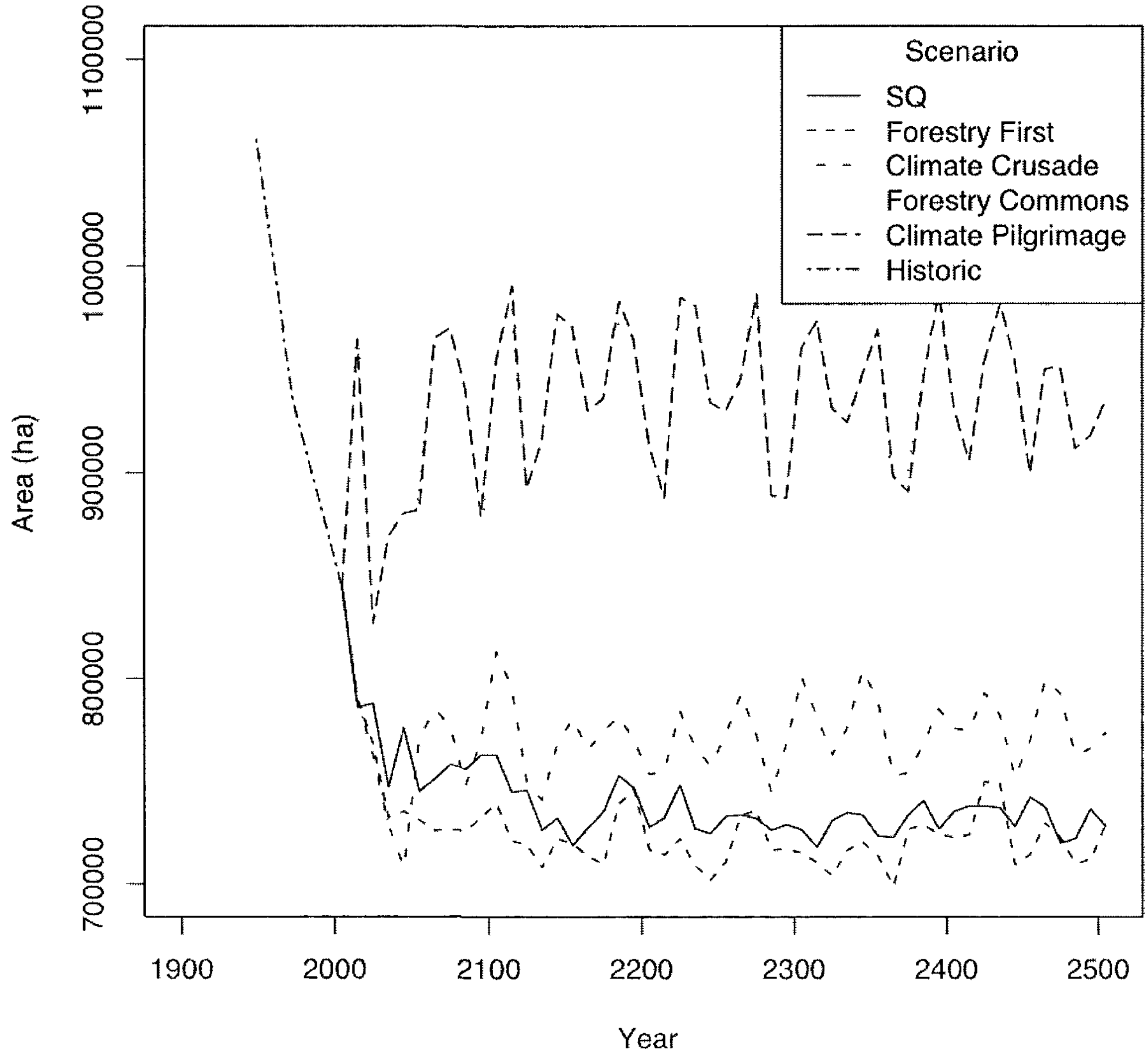

Figure 3-6. Total area of security patches over 10 replicates for the Status Quo, Forestry First, Forest Commons, Climate Crusade and Climate Pilgrimage scenarios over 500 years of simulation in the Cranbrook study area. As a comparison historic data for 1949 and 1973 are included.

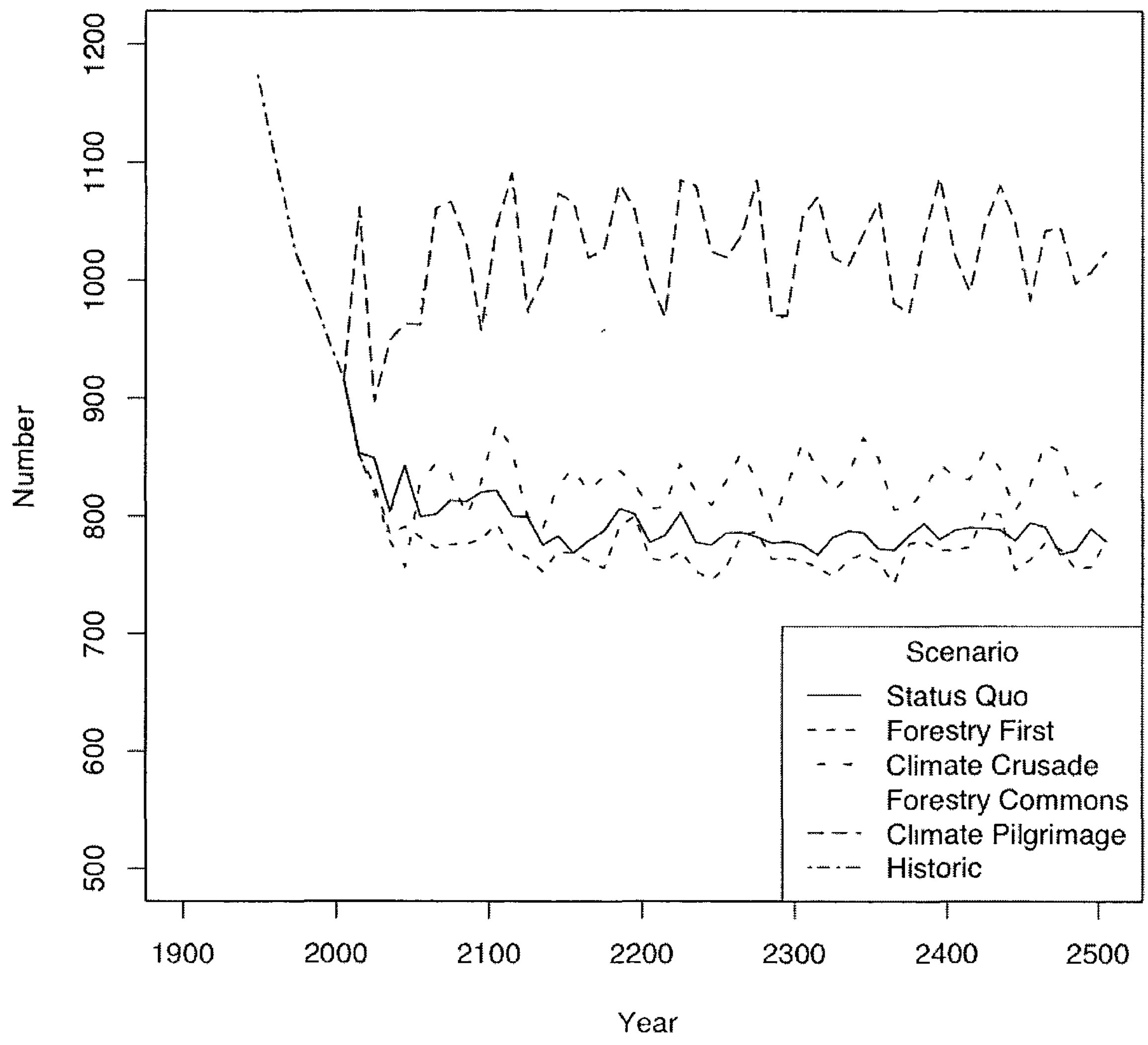

Figure 3-7. Mean number of patches secure from human access over 10 replicates for Status Quo, Forestry First, Forest Commons, Climate Crusade and Climate Pilgrimage scenarios over 500 years of simulation in the Cranbrook study area. As a comparison historic data for 1949 and 1973 are included.

Coarse-Filter Biodiversity - Regulatory Service

The Forestry Commons scenario most closely resembles the distribution of age classes of the No Management scenario, indicating that it is closest to natural landscape conditions. The 100 to 250 year age group of forest is most common in the No Management, Forestry Commons and Status Quo scenarios (21%, 19% and 17% of the total area, respectively; Figure 3-8) as compared to the Forestry First, Climate Crusade and Climate Pilgrimage scenarios (where it is projected to occupy only 14%, 13%, and 11% of the area, respectively). The Status Quo scenario had the most forest in the 250+ age class (15%) due to fire suppression. Despite having a low biodiversity emphasis, fire suppression also benefited the Forestry First scenario (14% of area in 250+ age class). The others are below the Status Quo and Forestry First scenarios (No Management 10%, Forestry Commons 10%, Climate Crusade 7% and Climate Pilgrimage 3%), with Climate Crusade fairing better under climate change due to fire suppression. The age distribution of the forest modelled in all scenarios follows a negative exponential distribution due to the actions of simulated fire and MPB. The Kolmogorov-Smirnov test found that the distribution of age class data generated for each scenario did not differ significantly when compared to the No Management scenario.

Although there are a total of 32 ecosystem groups in the study area, on average the No Management scenario only generates enough old forest for 35% of the units to be above their representative amount (Figure 3-9). The Status Quo scenario does the best with respect to mature forest representation (43%) due to fire suppression and a modest harvesting regime. The Forestry Commons (32%) scenario performs marginally better than the Forestry First (30%) scenario and is closest to the no management scenario, but without fire suppression it is far below the Status Quo scenario. Both climate change scenarios perform poorly, especially the Climate Pilgrimage scenario that only achieves 8%, whereas the Climate Crusade is at 19%. The variability in the occurrence of old forest is much higher in the absence of fire suppression for the No Management

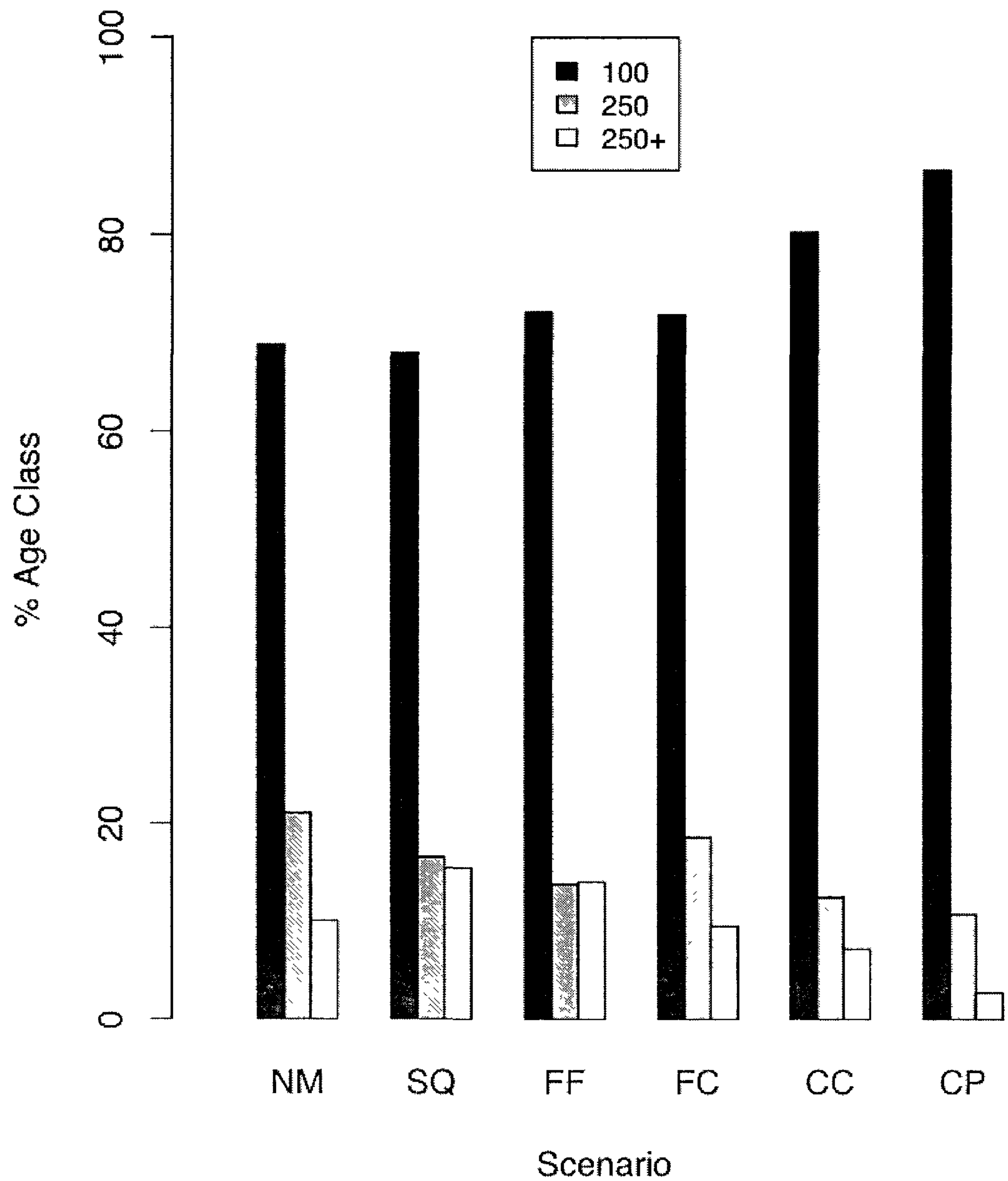

Figure 3-8. Mean percentage of hectares of forest in three different age groups (stand age ≤ 100 years, > 100 to ≤ 250, >250) over 10 replicates by each scenario at simulation year 250 over the study area (No Management, Status Quo, Forestry First, Climate Crusade, Forest Commons, Climate Pilgrimage).

(sd=5.7) and the Forestry Commons (sd=5.1). The variability is less for the Forestry First (sd=2.6), Climate Crusade (sd=3.4), and the Climate Pilgrimage (sd=3.5) scenarios. Fire suppression changes forest structure through the maintenance of old forest.

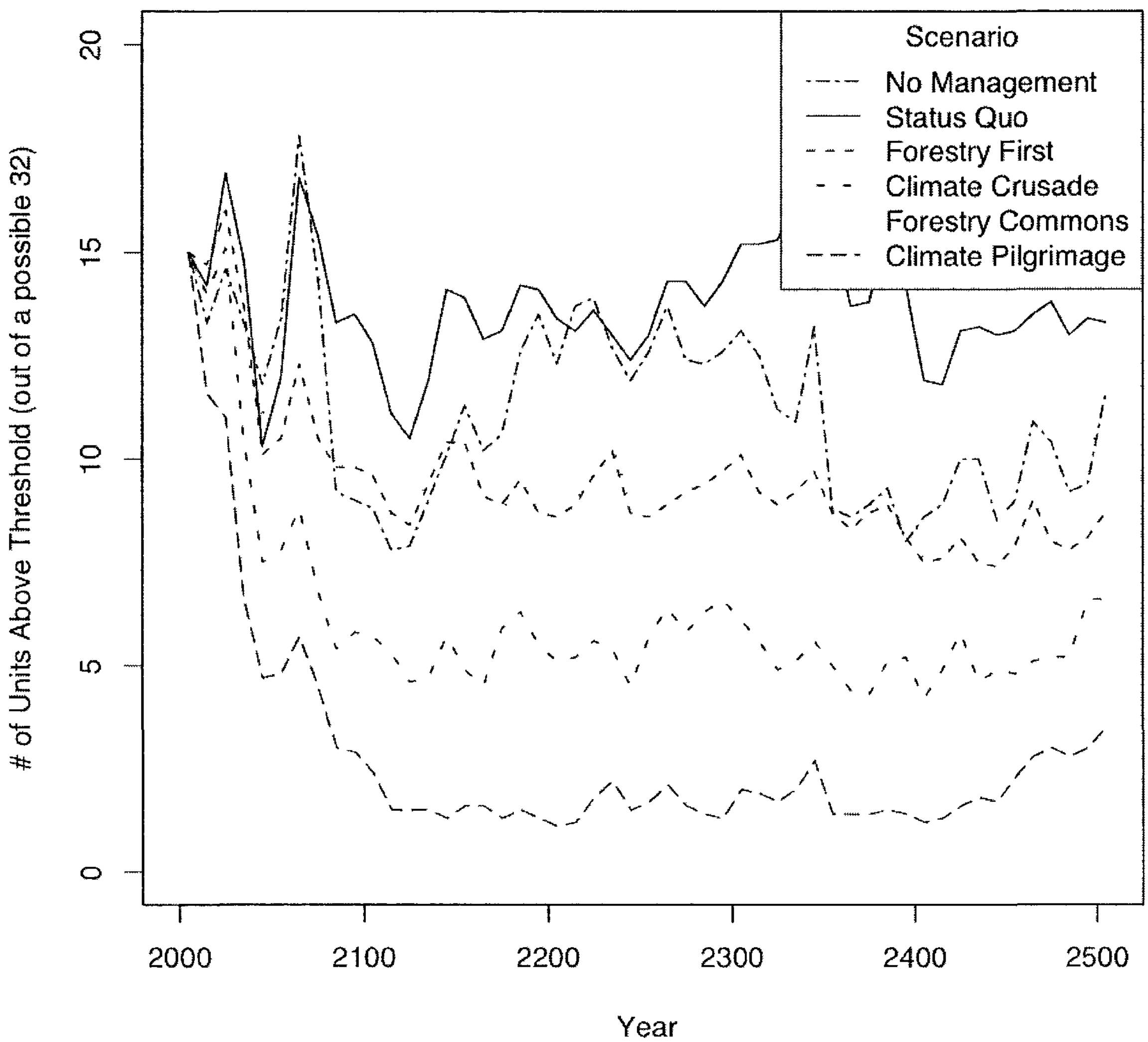

Figure 3-9. Mean number of ecosystem units (Appendix C) over their respective "old" threshold -- amount of old expected under historic disturbance regime -- in the Cranbrook study area. Results calculated over 10 replicates for No Management, Status Quo, Forestry First, Forest Commons, Climate Crusade, and Climate Pilgrimage scenarios for 500 years of simulation.

Under the No Management scenario, as expected, fire and MPB greatly influence the amount of area disturbed (Figure 3-10; 79% and 21% of all disturbance). Even with harvesting, but no fire suppression, the Forestry Commons scenario is also dominated by fire (65%). MPB is dampened in the Forestry Commons scenario presumably by forest harvesting and increased fire (13%). Although fire is still prevalent, harvesting represents an increased proportion of area disturbed for the Status Quo (fire 43%, harvest 41%) and Forestry First (fire 43%, harvest 43%) scenarios representing a lesser role of fire as a controlling process compared to the No Management regime. While MPB is initially high it does drop (under the Status Quo to 12%, Forestry First 9%), but large events do periodically occur. Fire rises for both the climate change scenarios, as expected, but without fire suppression it is most extensive in the Climate Pilgrimage scenario (Climate Pilgrimage 87%, Climate Crusade 68%). Under these two scenarios the incidence of MPB drops due to the increased frequency and extent of fire (Climate Pilgrimage 2%, Climate Crusade 3%).

Fire suppression has a large impact on a number of components of the system, including a greater area of old forest and a modest increase in MPB activity. The area of susceptible pine (Figure 3-11) rises from historic levels starting in 1973 then peaks and declines under all scenarios. The level of susceptible pine is highest and the most variable under the No Management and Forestry Commons scenarios ($\bar{x}$ = 115,447 ha, sd = 30,218 and $\bar{x}$ = 113,068 ha, sd = 23,820), in contrast to the Status Quo and Forestry First scenarios ($\bar{x}$ = 105,654 ha, sd = 16,856 and $\bar{x}$ = 100,960 ha, sd = 16,058). Under the No Management scenario there is no harvesting to decrease the area of old pine and it is modestly higher than historic levels. The variability of susceptible pine is highest for the No Management scenario (sd=23,820). The amount of susceptible pine drops under climate change, due to the presence of harvesting and a higher incidence

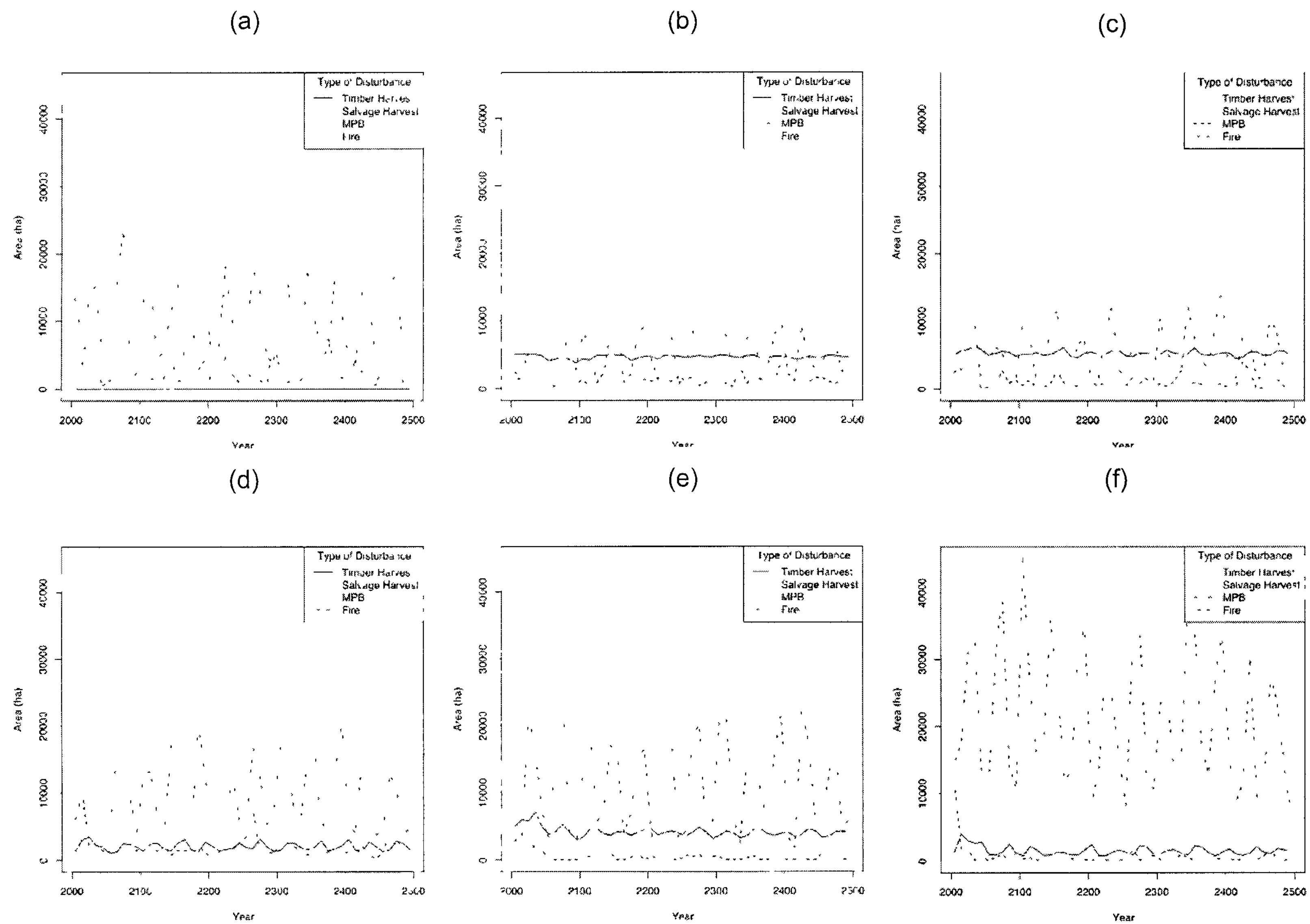

Figure 3-10. Mean area of disturbance type over 10 replicates and six scenarios for 500 years of simulation. No Management (a) Status Quo (b) Forestry First (c) Forest Commons (d) Climate Crusade (e) Climate Pilgrimage (f).

of fire for the Climate Pilgrimage and Climate Crusade scenarios ($\bar{x}$ =72,251 ha, sd=27,591 and $\bar{x}$ =80,935 ha, sd=19,188).

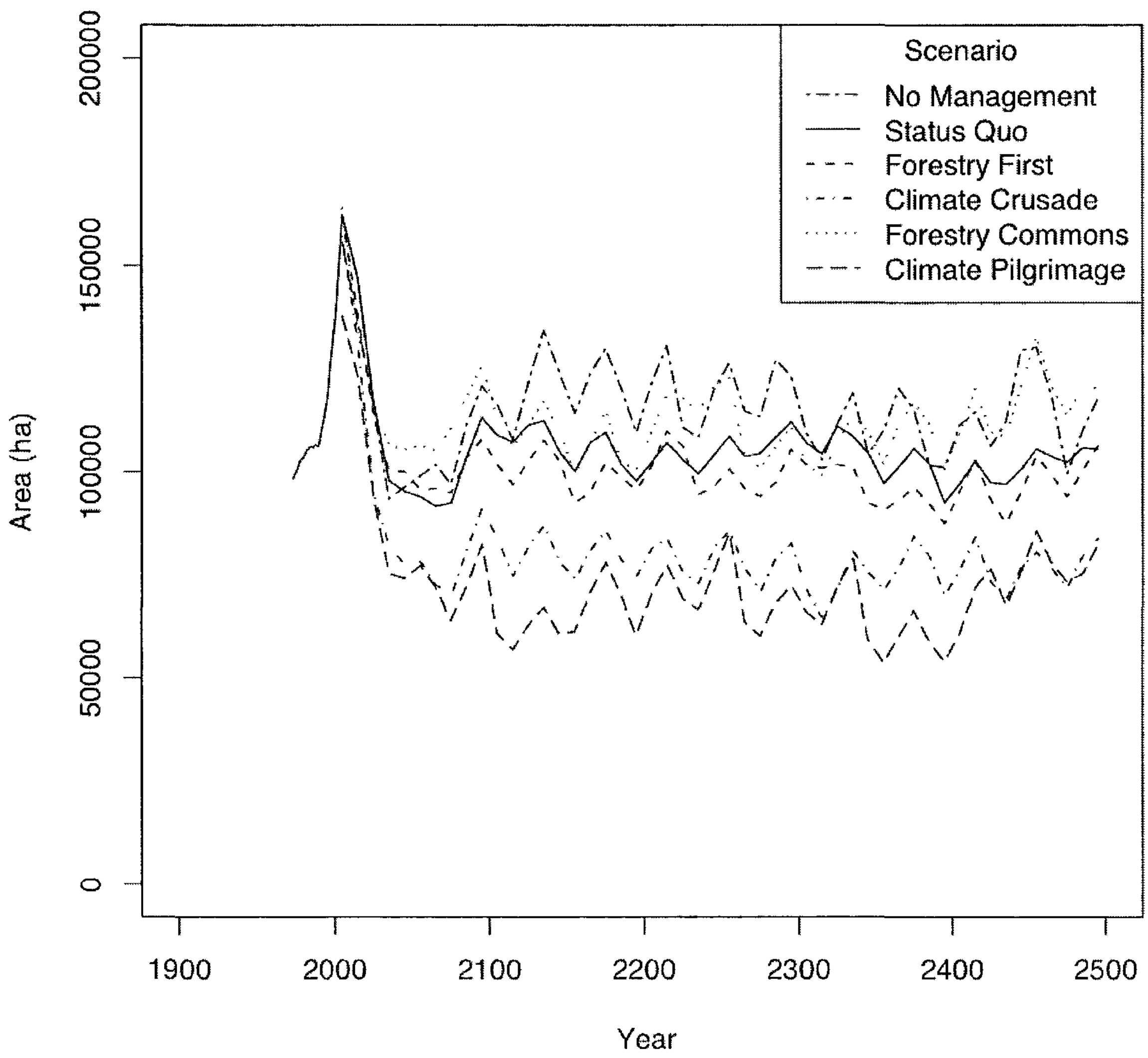

Figure 3-11. Mean percentage of susceptible pine by age-class over 10 replicates for No Management, Status Quo, Forestry First, Forest Commons, Climate Crusade and Climate Pilgrimage scenarios over 500 years of simulation.

Discussion

Using scenarios to explore how a forest may change under a range of social and ecological conditions provides insights into the interaction of management choices and ecological processes (Peterson et al. 2003). Some simulation outcomes were expected on the basis of assumptions about system behaviour. Examples include the amount of forest burned, timber targeted to harvest, and the triggering of MPB events with increasing amounts of susceptible pine. Other interactions were unexpected, emerging through the analysis. These included the interaction of salvage with disturbance, the role of a 'salvage-only' zone in augmenting harvest levels, the implications of aggressive harvesting for managing regulatory and provisioning services, and the greater amounts of old forest that occurred under some scenarios. Also unexpected was the increased amplitude of all indicators without fire suppression and in the presence of climate change. A steady growing stock is a key objective of current management; however, under all scenarios growing stock oscillated. This demonstrates the contradiction of trying to manage a dynamic system as if it was stable. With periodic extensive disturbance it is inevitable that growing stock will rise and fall and management should be structured to accommodate this dynamic.

Only the passive scenarios explicitly incorporated disturbance into timber supply planning. These scenarios were able to exceed their base harvest request by responding to salvage opportunities and managing timber supply dynamically rather than uniformly. The Status Quo and aggressive approaches to forest management, operating under an assumption that future disturbance would be manageable, failed to meet their intended harvest goals, this despite the benefit to the aggressive scenario of a larger THLB and relaxation of conservation objectives. Further, the Status Quo and aggressive approaches did not provide an expected consistent timber supply, because of the interaction of disturbance, salvage and over harvesting generating an oscillating growing stock.

The Status Quo scenario performed relatively well for both timber supply and old forest representation. Given the reduction in services under Climate Crusade and Climate Pilgrimage it is likely that the Status Quo scenario would suffer a similar drop in the future with climate change. With the exception of old forest representation, the Forestry First scenario only performed marginally poorer than the Status Quo for the indicators assessed.

The regulatory ecosystem services, grizzly bear natal habitat, the distribution of forest ages, amount of old forest, the dominant disturbance process, and the amount of pine susceptible to MPB, give a broad representation of the state of the resource system. By tracking regulatory services, resource professionals can identify the threshold at which the forest will shift to a different ecosystem state. For example, an increase in susceptible pine strongly indicates a loss of resilience to MPB. Shifts in forest age structure and dominant disturbance agent indicate that there has, at least at one scale, been a change in regime given that the controlling processes have shifted from fire to forest management. Recently, as throughout much of interior BC, there has been a decline in resilience to MPB across the Cranbrook study area. Fire suppression has led to an older cohort of pine than would not have been expected under the historic disturbance regime (Taylor and Carroll 2004). However, this is likely to be dampened in the future by either an increase in fire or further conversion of the forest to management. This interpretation agrees with other findings (Taylor and Carroll 2004) that indicate that the recent increase in susceptible pine may be temporary and as the pine is harvested and the stand age is kept below 100 year, the likelihood of future MPB outbreaks will decrease. It would appear that the current MPB event is a product of a state transition from an unmanaged to a managed forest and large historical fires.

When systems change states there is a possibility of a period of chaos as new controlling processes become established (Walker and Meyers 2004, Scheffer et al. 2009). Boundary chaos clouds the identification of a definitive threshold between alternative states. Transition chaos will become more of a concern as ecosystems re-organize due to climate change. More research will be required to detect early warning signals of potential regime shifts in forested ecosystems (Scheffer et al. 2009).

Based on the scenarios I assessed, fire suppression contributed to an increase in old timber for harvest and in old forest for conservation. There are mixed advantages to fire suppression: for example, with more old forest there is more habitat for old forest dependent species, however this could be at the detriment of early seral species (Bunnell 1995). Skewing forest age structures, away from what would have occurred under natural processes, could undermine the capacity of the broader ecological community to respond to disturbance events (Drever et al. 2006).

Fire suppression has been effective in the past (Daniels et al. 2007) and it is reasonable to assume that it would be in the future. However, based on modelling studies, climate change is anticipated to increase annual area burned by 100%. Given the past success rate of suppression (an approximately 50% reduction in annual area burned; DeWilde and Chapin 2006), the future forest will be more dynamic and perhaps comparable to current boreal landscapes that do not have significant fire suppression. As a result, future landscapes will need to be managed under an expectation of greater amounts of fire disturbance. This will inevitably lead to spikes in timber availability as salvage opportunities emerge. There are, however, substantive ecological and management issues associated with fire suppression. Fire suppression, for example, can reduce structural complexity (Bergeron et al. 2002, Kuuluvainen 2002, Drever et al. 2006, Puettmann et al. 2009). The simplification of forest structure can reduce resilience to a range of disturbance types; for example a build-up of fuels can trigger extremely large fire events (Arno et al. 2000) and the lack of regeneration of some fire-dependent forest species (Zackrisson 1977). Implementing measures to support diverse forest structures and minimize fuel loads, such as the use of prescribed burning, would help address some of fire suppression's limitations (Lindenmayer et al. 2008).

Homogenizing temporal dynamics is intended to provide a consistent supply of ecosystem services. The scenarios with fire suppression have more mature and old forest. This shift in age structure can lead to a loss of resilience and trigger a large-scale event, such as the recent MPB outbreak in BC. Dampening temporal dynamics may be the ultimate trade off between maximizing timber production and the encouragement of a more natural age structure necessary for maintaining resilience to large-scale disturbance events. By using strategies such as leaving sections of the landscape unmanaged or using a salvage only zone, timber can be harvested in a way that is more cognizant of landscape processes. Furthermore, a dynamic approach to managing timber supply can provide better protection of other values, such as grizzly bear habitat.

Managing for resilience entails maintaining biodiversity and allowing for spatial and temporal ecological dynamics.

The scenarios I developed reflect different perceptions of risk associated with future events. The passive scenarios are more risk averse, with a smaller land base and an adaptive response to perturbation. The more aggressive approach assumes a stable, predictable future with an increase in harvest as a response to disturbance. This harvest response leads to near-term shortfalls in timber supply and biodiversity, and has implications for forest structure, composition and pattern. This dynamic behaviour becomes more pronounced under the climate change scenarios due to the increased rate and extent of disturbance. Economically, there may be less risk in the short term to maximize harvest, due to economic discounting (Chapin and Whiteman 1998), however this approach leads to a trade-off of short-term economic benefit with longer-term economic and ecological risk. This trade-off would best be addressed through public debate on the short- vs. long-term risk and sustainability of ecosystem services.

Managing for resilience includes strategies to promote structural, compositional and pattern diversity, along with approaches to support ecosystem dynamics and adaptation to extreme events (MA 2005, Drever et al.2006, Millar et al. 2007, Campbell et al. 2009). Table 3-4 contrasts the four scenarios with respect to diversity (table headings Biodiversity, Structure, Pattern, and Composition), how each approaches dynamics (Managing Dynamic, Disturbance Assumption, Ecological Process, and Extreme Events) and promotes adaptation to extreme events (Planning for Change, Key Conservation Features, Redundancy, and Connectivity). Each scenario is summarized according to these broad categories (Diversity, Dynamics and Adaptation). The passive

Table 3-4. Summary of four scenarios and how they provide resilience to disturbance, manage dynamics and promote adaptation to landscape events (-- strategy not met, - strategy moderately not met, + strategy moderately met, ++ strategy met). A resilience summary is given in the last row.

Strategy	Result	Forestry First	Forestry Commons	Climate Crusade	Climate Pilgrimage
Management Objective		Maximum sustainable yield. More old forest due to fire suppression	Increasing functional and response diversity - limited success due to increase in fire, but similar to FF	Minimize loss of timber to disturbance. Loss of old forest due to increased fire	Increasing functional and response diversity - poor success due to increase in fire
Biodiversity		Apply low biodiversity landscape objectives over landscape	Apply mix of low, medium and high biodiversity landscape objectives	Apply low biodiversity landscape objectives over landscape	Apply mix of low, medium and high biodiversity landscape objectives
	Old forest representation	+	+	-	--
Structure		More uniform age structure, harvest at cumulation age, negative exponential over all	Age structure consistent with disturbance regime	Negative exponential age structure due to increased incidence of fire	Age structure consistent with disturbance regime
	Negative exponential - age structure	+	++	++	++
Pattern	Harvesting - 20-60 ha blocks. Fire - 1,000 ha MPB - 4 ha	Harvesting dominated - smaller openings	Fire dominated - larger openings	Harvesting dominated, but with more fire - more larger openings than FF, but less than FC and CP	Fire dominated - larger openings, more than FC
	Variable sized disturbance patches	--	+	-+	++
Composition		Monoculture - most disturbance harvest, then replant	Mixed - most disturbance fire, natural regeneration, some replant	Monoculture/Mix - disturbance mix of harvest and fire	Mixed - extensive fire, natural regeneration, some replant
	Tree species diversity	--	+	-	++

Strategy	Result	Forestry First	Forestry Commons	Climate Crusade	Climate Pilgrimage
Managing Dynamic		Maximize harvest, minimize loss to disturbance	Bet-hedging - anticipate future unknowns	Maximize harvest, minimize loss to disturbance	Bet-hedging - anticipate future unknowns
	Loss to disturbance	-	-	--	--
Disturbance Assumption		Impacts averaged and assumed on an annual basis	Focus on variation and anticipate periodic large scale salvage	Impacts averaged and assumed on an annual basis	Focus on variation and anticipate periodic large scale salvage
	Disturbance anticipation	-	++	--	+
Ecological Process		Extensive fire suppression	No fire suppression, fire consistent with historic	Extensive fire suppression, less success than FF	No fire suppression, extensive fire
	Fire suppression success	++	-	+	--
Extreme Events		Assume consistent supply, if extreme event occurs then redo management plans.	protocols for response to periodic large scale events	Assume consistent supply, if extreme event occurs then redo plans.	protocols for response to periodic large scale events
	Plan for extremes	-	++	--	+

Strategy	Result	Forestry First	Forestry Commons	Climate Crusade	Climate Pilgrimage
Planning for Change		No anticipation. increase harvest in response to disturbance	Salvage only zone. When disturbance increase harvest into zone.	No anticipation. increase harvest in response to disturbance	Salvage only zone. When disturbance increase harvest into zone.
	Anticipating disturbance	-	++	-	++
Key Conservation Features		Violate conservation constraints to meet timber targets	Maintain high conservation areas	Violate conservation constraints to meet timber targets	Maintain high conservation areas, salvage only zone harvested more heavily with increased disturbance
	Conservation	-	++	--	+
Redundancy		Single ecological representation and some areas can be harvested	More area excluded from harvest and use of no harvest buffers around conservation areas	Single ecological representation and some areas can be harvested	More area excluded from harvest and use of no harvest buffers around conservation areas
	Replication in conservation	-	++	--	+
Connectivity		In non harvesting land base only. increased industrial activity	Multiple connections at multiple scales. access constraints. Limited industrial activity	In non harvesting land base only. increased industrial activity. but concentrated due to CC salvage	Multiple connections at multiple scales, access constraints. Limited industrial activity and concentrated
	Areas secure from humans	--	+	-	++
Resilience Summary					
Diversity		-	+	-/+	++
Dynamics		-	++	--	+
Adaptation		-	+	-	+

scenarios have the most resilience management features because they anticipate and manage dynamics. The aggressive scenarios are limited in managing for resilience due to their focus on maximum sustained yield rather than promoting flexibility to deal with future events.

The difficulty of managing for any large unexpected disturbance is a product of our lack of understanding about system boundaries and how to develop sustainable practices that work in a local context. Also, identifying the position of the system relative to an alternative state is key to better managing for large-scale disturbance events. This requires research, biophysical monitoring, regular assessments, education and balancing social expectations with the capacity of systems to provide services, and the regularity of their supply as shown in this project.

A forest management planning approach that focuses on resilience and incorporates protocols of response to disturbance, like no salvage in key conservation areas and shifting areas for industrial use or habitat conservation, would be more appropriate for dynamic systems such as those anticipated with a changing climate (Rayfield et al. 2008). Maintaining a network of connected conservation areas would provide more flexibility in the event that parts of the landscape become heavily disturbed (Eng 1998). Extreme events would be tempered by ensuring complexity, and enhancing response diversity, to promote post-disturbance re-organization. This approach should intentionally promote management flexibility, and the acceptance of future disturbance events and the associated salvage opportunities that would accompany a fluctuating availability of timber (Chapin and Whiteman 1998, Lindenmayer et al. 2008).

Several improvements in the simulation model would provide deeper insights and a more detailed assessment of stand- and landscape-scale ecological dynamics. The model I developed used only one set of post-disturbance successional pathways for each stand type. A broader range of pathways, linked to the extent and type of successive disturbances would provide a better representation of stand-scale processes (Frelich and Reich 1998). As well, insights would be gained into how different stand responses influence resilience. The model lacked clear feedback mechanisms between management approaches such as fire suppression and harvesting pattern and how this interaction might influence resilience. The modelling of climate oscillation was on a regular interval; further analysis of historic trends may have provided insights into the variability of ocean-atmospheric phenomena which could have been incorporated into the model. The fire model could be improved by introducing some process-based features such as an increase in fire initiation and volatility with an index of fuel loads linked to a stand's disturbance history.

The scenarios provide context for informing forest management and how current approaches should be structured to plan for inevitable dynamics. Managing for resilience is not about implementing some new forest management prescription, but rather about building a flexible approach to managing resources, such that uncertain events can be tempered and are incorporated into forest management planning. Shifting to managing for resilience requires social shifts as well. People's expectations and assumptions to how and when forests can supply services would need to be re-thought to adapt to managing for a dynamic resource.

Disturbance events are a required component of healthy forests and therefore are necessary to ensure the continuation of ecosystem services (Puettmann et al. 2009). Disturbance cannot be avoided; the unknown is when, where and how much disturbance will occur. Scenarios help to explore the extent, frequency and impacts of future disturbances and the interaction with different approaches to the management of ecosystem services such as the role of fire suppression and the implementation of a salvage-only zone. Looking at a range of scenarios, as opposed to one preferred future, provides greater insights into current management and opportunities for adapting to the future, especially when trying to balance biodiversity and timber supply, while dealing with social and ecological uncertainty.

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CHAPTER 4

Conclusion

The capacity for ecosystems to consistently supply ecosystem services has become undermined by ecological degradation and an increased incidence of natural disturbance events (Carpenter 2003, Walker and Meyers 2004, Adger et al. 2005, Folke et al. 2004, MA 2005, Hobbs et al. 2006, Williamson et al. 2009). The frequency and areal extent of disturbance is expected to increase as the climate continues to change (Emanuel 2005, IPCC 2007). Expectations of the reduction in ecosystem services are based on historic occurrences of natural disturbance events. Current approaches to resource management are not well suited to deal with predicted increases in disturbance (Spittlehouse and Stewart 2003, Lindenmayer et al. 2008, Williamson et al. 2009). My research goal was to apply and extend methods for evaluating social-ecological systems as a way of addressing the limitations of conventional resource management approaches.

Spatial and temporal dynamics play a vital role in generating the structural and compositional complexity in ecosystems that enables natural systems to adjust to shifting climate and natural disturbance (Scheffer 2001, Gunderson and Holling 2002, Drever et al. 2006, Puettmann et al. 2008, Campbell et al. 2009). Social-ecological systems theory emerges as a theoretical basis for developing resource management approaches that recognise and incorporate spatial and temporal dynamics. Further, social-ecological approaches integrate the needs of people relative to ecosystem services and the dynamic environment that provides them (Gunderson and Holling 2002, Walker et al. 2004, Walker 2005, RA 2007).

Social-ecological theory focuses on a system's resilience, adaptability and transformation (Walker et al. 2004). Resilience is the capacity of a system, such as a forested ecosystem and its management, to persist after disturbance and still maintain its defining characteristics and processes (Holling 1973, Carpenter et al. 2001). The adaptability of a social-ecological system is tightly connected to resilience. Where resilience relates to the capacity of the system to persist, adaptability links to the social and ecological mechanisms of that persistence. Social adaptive capacity embodies institutional flexibility, technical innovation, and social networks, all of which increase people's capacity to respond to change (RA 2007a, 2007b). Ecological adaptive capacity correlates to response diversity, an important feature of ecological complexity (Campbell et al. 2009, Puettmann et al. 2009).

When the social or ecological capacity of the system is overcome, or when the current configuration is shown to be no longer appropriate, transformation occurs. Transformative change is now necessary to adapt to climate change. New resource management approaches are required to address the social expectation of resource extraction and to sustain ecological capacity for future generations (Pojar 2010).

In Chapter 2, I used a structured framework to examine an alternative approach to resource management. By assessing resilience, adaptability and possible transformation, the framework evaluates a resource system's historic, current and possible future configurations, and provides insights into an ecosystem's adaptive capacity. Using this framework, communities and resource managers can conduct social-ecological assessments allowing them to anticipate transformation and take the steps necessary to mitigate the negative consequences.

Understanding future social and ecological dynamics is critical to planning for a supply of ecosystem services. Typically, a conventional resource management plan assumes one future, then builds resource allocation models to determine levels of harvest. Sensitivity analyses are conducted to understand the implications of variation in input assumptions (Province of BC 2007). When extreme disturbance events occur, the current resource models incorporate the losses and a long-term sustained yield is re-established (Province of BC 2007). However, the assumption of a stable future is not re-evaluated. The problem of not representing the full extent of system dynamics is apparent as forests are becoming more heavily influenced by a changing climate (Williamson et al. 2009, Pojar 2010).

Using a structured approach to build a range of scenarios depicting a representation of the future is a robust technique for understanding the social and ecological drivers of change (Peterson et al. 2003, MA 2005). Scenarios help us to understand the dynamics and uncertainty associated with the interaction and evolution of parts of a system. The methodology for composing structured scenarios, shown in Chapter 2, divides the range of possible futures by their social and ecological components. Each scenario was structurally unique. The rate and extent of natural disturbances were featured on the ecological axis. The rate of disturbance was either similar to historic, or more extreme, driven by changes in climate. Socially, the scenarios ranged from a passive to an aggressive approach to forest management. The passive approach attempted to recreate historic landscape dynamics, whereas the aggressive was parameterized to maximize timber harvesting at the detriment of other values. There are an infinite number of possible futures; however, by entertaining four options across a gradient of possibilities, insights are gained into the emergent properties of dynamic resource systems. The construction of scenarios helps to identify the amount of ecosystem services that can be expected given the fluctuations in resource supply.

Qualitative scenarios allow resource professionals to compile information on a resource system and to engage interest groups in developing a common understanding of what has historically occurred, and how it influences the present. When attempting to peer into the future, we generate questions that help us understand uncertainty: to what extent may events play out; and how are they buffered or encouraged through the interaction of social and ecological processes? By quantifying the scenarios, and conducting simulation experiments, as presented in Chapter 3, some of these questions are addressed. The process of generating models forces us to understand the state of our knowledge relative to the behaviour and interactions of ecological and social processes (Starfield 1997). Model predictions provide some insights into the future; however, the scenario process guards against interpreting specific outcomes and instead forces us to look at differences in outcomes relative to the suite of scenarios that are identified.

The Cranbrook management unit in southeastern BC was used as a case study to demonstrate the scenario analysis methodology. A set of simulation models was constructed, using the SELES (Fall and Fall 2001) spatio-temporal modelling tool, that captured landscape dynamics and forest management. A number of the assumptions within the scenarios were confirmed. For example, harvesting aggressively compromises fine- and coarse-filter biodiversity, and forgoing fire suppression and reducing the timber harvesting land base severely restricts annual timber supply. Other properties of the system were not originally anticipated, particularly the interplay of system dynamics and the supply of ecosystem services. Aggressive forest management has difficulty maintaining a consistent supply of timber when landscape dynamics change. The oscillation in timber availability was unintended and emerged because of the strategy to constantly maximize the amount of timber harvested used in the aggressive approach. Socially, this would be very disruptive, as expectations would be built for a certain level of harvest only to be compromised by sudden declines in availability of timber. The more passive set of scenarios relied on a smaller land base for harvest and used an opportunistic salvage zone. Under this class of scenarios, social expectations could be built around a lower level of consistent economic activity. However, an assumption of the passive approach is that disturbance occurs periodically and that harvesting increases temporarily in response.

The aggressive and passive approaches to resource management reflect fundamental differences in philosophy. The aggressive approach is attempting to control nature to provide services at an annual scale that meets human need for consistency and certainty (Chapin and Whiteman 1998). By anticipating disturbance and adapting, the passive approach is more inclined to work with natural dynamics. This approach accepts oscillation in the availability of ecosystem services as part of the nature of the system. In the end, the passive approach is able to preserve values besides timber, yet still harvest a substantial amount of timber by working with, rather than against, natural dynamics. Ultimately, and despite our goal of stability, natural disturbance and resource extraction results in the oscillation of ecosystem services such as timber supply.

Resource professionals have a choice: to incorporate the inevitability of future dynamics into planning and management, or not. For the Cranbrook study area a combination of management strategies from each scenario would yield the most robust approach to managing for social-ecological resilience. For example, monitoring the supply of ecosystem services and implementing strategies to dampen their oscillation (fire suppression) or to take advantage of resource pulses (salvage) would be the most social-ecologically appropriate.

Resilience is a broadly applied concept that is focussed on encouraging complexity and adaptive capacity in social-ecological systems (MA 2005, RA 2007a, Puettmann et al. 2009). For forested systems at the stand level, this means advocating for management practices that: prioritize stand diversity; maintain biological legacies; promote heterogeneity across a forest; encourage diverse species mixes for replanting; and harvest stands at different culmination ages. Similarly, at the landscape scale, a resilience approach entails diversifying harvest unit size and spatial arrangement, varying cutting strategies and implementing prescribed burning as a different approach to fire suppression, in order to maintain ecological processes (Campbell et al. 2009).

Temporally, resilience is about working with the disturbance dynamics of the system. This requires a social shift to accept uncertainty and recognize that a larger component of harvesting will be opportunistic. As well, taking advantage of salvage opportunities may entail constraining human activity. For example, an extensive road network would be required to salvage disturbed forest, which may require strict access control to manage for other values, such as grizzly bear (*Ursus arctos*) natal areas. Following the principles of resilience theory, managing a "portfolio" of forests is the most appropriate strategy. When a forest is disturbed some parts should be salvaged, some left alone, while undisturbed areas are harvested at a far lower rate. With climate change expected to increase the area burned across western North America by 100%, resilience approaches are key to ensuring a future supply of ecosystem services (Wotton and Flannigan 1993, Stocks et al. 1998, Li et al. 0 Flannigan et al. 2005, Nitschke and Innes 2008, Krawchuk et al. 2009). Although the supply of services may not be steady or stable, a resilience approach that uses scenario modelling to manage for a range of possible futures will enhance the chances for functional ecosystems and as a result human well being.

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Appendix A. Summary of Fire History Analysis for Implementing Fire Modelling in the Cranbrook Study Area.

I used historical fire data provided by the Canadian Forest Service to model to model future fires across the Cranbrook study area. The data included date of ignition, ignition sources and area burned for 682 fires recorded or inferred between 1919 and 2000 (Taylor pers. comm.). Over that time period 778,695 ha of forest burned. For the fire model these data were analysed to calculate disturbance rates and patch sizes for each NDT in the Cranbrook TSA.

The fire data had a skewed distribution (Figure 1). When only the smaller area disturbances were considered, below the 95th percentile (fires < 4,554 ha in size; methodology after Morgan et al. 2008), a total area of 260,396 ha (658 fires) was disturbed by fire. The 34 large fires, above the 95th percentile (Figure 1), accounted for 518,334 ha of area burned. The majority of the large fires occurred prior to 1940 (33 of 34), before large-scale fire suppression occurred across the study area. However, some researchers have reported a correlation between the warm phase of the Pacific Decadal Oscillation (PDO) and large regional fires (Daniels et al. 2007, Morgan et al. 2008). To investigate the role of the PDO in the Cranbrook, I partitioned the fire data into warm (45 years; 1925-1946, 1977-1998) and cool (37 years; 1919-1924, 1947-1976, 1999-2000) PDO phases (University of Washington 2010). Both large and small fires were highly correlated with the warm phase of the PDO, with 27 of the 34 large fires and 413 of the 658 small fires occurring in those years. This result supports the hypothesis that large fires in the Cranbrook were correlated with the PDO warm phase.

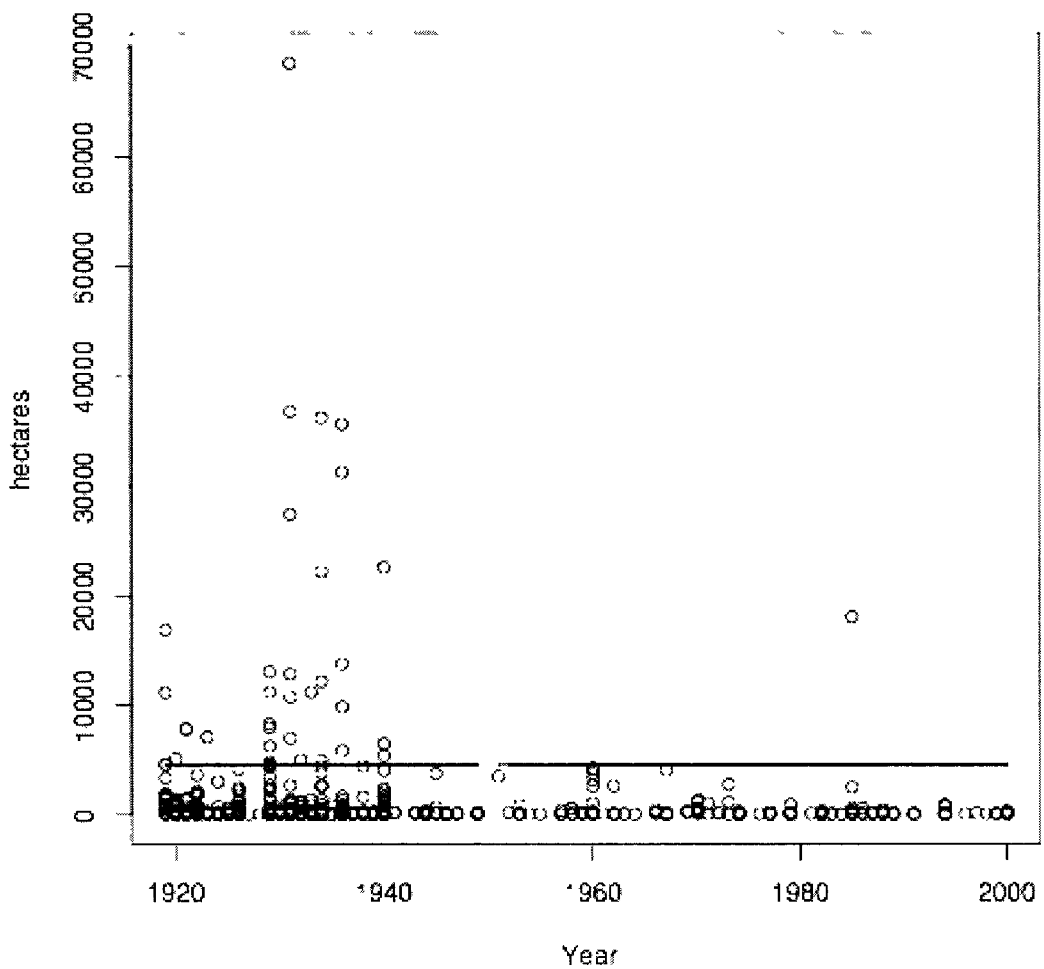

Figure A-1. Plot of fire frequency by size and year for the Cranbrook study area. Black line indicates 95th percentile of fire size, fires above line are greater than 4554 ha.

In order to increase the resolution of the fire model the fire data were partitioned spatially according to their natural disturbance regime, termed here as natural disturbance type (NDT; Province of BC 1995). Some fires straddled NDTs. In these cases, the historic fires were allocated to NDTs based on the largest NDT/fire overlap. The fire return interval (Van Wagner 1987) - the number of years to burn an area equal to the size of the forest - was an input parameter of the fire model and calculated for each NDT and PDO phase combination (Table A-1). The return interval (RI) was calculated as:

RI = NumYears x TotalArea/AreaBurn

where

NumYears is the number of years for which data were collected

TotalArea is the total area of the forest

AreaBurn is the total area burned over the years for which data were collected

Table A-1. The forested area, area burned and fire return intervals for the three main natural disturbance units in the Cranbrook study area.

	Natural Disturbance Unit		
	2	3	4
Forested area	104,896	770,958	161,755
Total area burned over 82 years	34,383	471,448	143,044
Area burned warm phase (45 years)	27,063	460,646	109,974
Return Interval - warm phase	174	75	66
Area burned cool phase years (37 years)	7,320	135,555	143,044
Return Interval - cool phase	530	210	42

By further summarizing the historic data I generated an additional set of fire parameters were assessed for each NDT/PDO phase. The PDO cool phase fire parameters are shown in table A-2 and the warm phase in table A-3.

Table A-2. Fire modelling parameters for PDO cool years for the Cranbrook study area.

NDT	Year with Fire out of possible 37	Number of Fires when fires	Mean Fire size when fires	Mean Area Burned when fires
2	8	1.63	563	7,320
3	28	4.75	1,019	135,555
4	24	4.46	309	33,070

Table A-3. Fire modelling parameters for PDO warm years for the Cranbrook study area.

NDT	Year with Fire out of possible 45	Number of Fires when fires	Mean Fire size when fires	Mean Area Burned when fires
2	15	2.27	796	27,063
3	28	8.14	2,020	460,974
4	35	5.09	563	109,974

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Appendix B. Summary of Historic and Downscaled Provincial Mountain Pine Beetle Data for Implementing Outbreak Modelling in the Cranbrook Study Area.

I used historic Mountain Pine Beetle (MPB) data provided by the Canadian Forest Service to model future outbreaks in the Cranbrook study area. The data included annual number of MPB outbreaks, their severity and their areal extent from 1935 to 1996 (S. W. Taylor pers. comm.). Each outbreak was classified according to severity: 1 (1 to 10% of stands attacked), 2 (11 to 30% attacked) and 3 (over 30% attacked). The modelling for this project is focused at the landscape scale and only explicitly considers stand initiating disturbance. I assumed that only severe attack (severity code = 3) would produce a significant level of stand mortality and that lesser values would produce stand complexity. As a result, calculations of the historic number and size of MPB events were based on outbreaks with a severity code of 3.

The historic MPB data were analysed for the number of patches, patch size and overall areal extent of outbreaks. For the Cranbrook timber supply area there were 67 years with data, of those 22 had MPB outbreaks. The number of MPB outbreak patches per year varied between 1 and 1,392, with an average of 472 (sd=432). The frequency distribution of outbreaks followed a negative exponential distribution. The size of the outbreak patches varied between 0.001 hectares and 1,946 hectares, with a mean size of 4 hectares (sd = 33). The distribution of outbreak patch sizes also followed a negative exponential distribution. The total annual area of MPB outbreak varied from 9 hectares to a maximum of 14,269 hectares ($\overline{x}$ =1,819, sd=3739; Figure B-1).

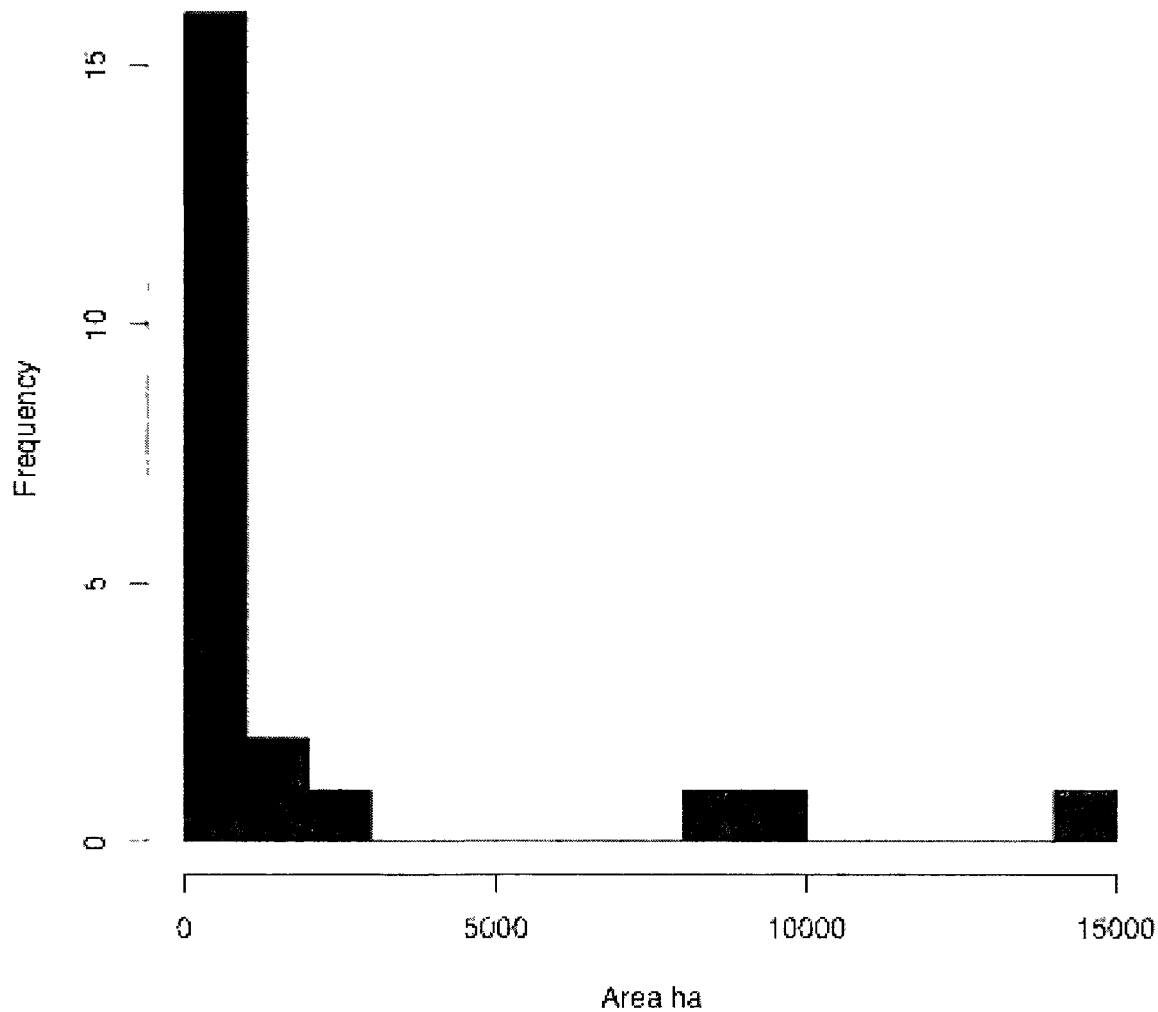

Figure B-1. Frequency histogram of area of historic Mountain Pine Beetle outbreaks for the Cranbrook study area.

I used data generated from a provincial MPB project (Walton et al. 2007) to model future outbreaks. Data from this project were downscaled to the Cranbrook Timber Supply Area. The provincial MPB forecast covers 27 years from 1999 to 2025 and reported a minimum outbreak size of 460 ha for the Cranbrook Timber Supply Area, a maximum size of 35,262 ha and a mean size of 8,201 ha. In combination, the historic data and the provincial modelled data generated a mean outbreak size of 5,276 ha; I used this value to represent the size of future outbreaks.

MPB outbreaks have occurred every 28-53 years in central BC (Alfaro et al., 2004, Taylor et al. 2006). In the Cranbrook study area or TSA MPB outbreaks were absent in the historic data prior to 1969, though present to the north in the Invermere Timber Supply Area dating back to 1930. As well, forest reports dating back to the early twentieth century recorded MPB outbreaks in the Elk (Hodgins 1931) and Flathead (Andrews 1930) valleys.

I used historic data to calculate the return interval for MPB across the Cranbrook study area. From 1945 till the end of current outbreak in 2025 an area of 245,015.8 ha is projected to be impacted by MPB resulting in an area of 306,314.8 ha impacted per 100 years. The area of pine is 374,903 ha resulting in a return time of 122 years under MPB alone.

The overall annual outbreak and outbreak patch size were found to follow a negative exponential distribution, however the shape of the distribution reflected a few large episodic events and smaller more typical events (Figure B-1). An MPB outbreak requires not only the availability of mature pine, but also favourable climate and dispersal conditions. When these factors converge a large-scale event is possible that is an order of magnitude larger than more typical outbreaks, such as the current MPB outbreak. To capture this effect the data were split at the 90th percentile, following a similar methodology used by Morgan et al. (2008) for differentiating large regional fires.

Table B-1 shows the mean outbreak size from the two groups. Epidemic outbreaks were assigned the mean from the data in the 90th percentile, while the size of endemic outbreaks was calculated from the remainder of the data.

Table B-1. Summary statistics for Mountain Pine Beetle outbreaks in the Cranbrook study area calculated from the historic and a combination of historic projected outbreak data.

	Years	Mean outbreak size	Mean Patch Size	Mean Number of Patches per Outbreak
Historic Data	22	1819	4	472
Historic + BCMPB	42	5276	NA	NA
Endemic outbreak	37	2878	NA	NA
Epidemic outbreak	5	27630	NA	NA

Based on the historic data, small endemic outbreaks initiated in the Cranbrook study area only after the amount of susceptible pine rose above 50,000 hectares and became common after 90,000 hectares. Epidemic outbreaks initiated when 90,000 hectares of susceptible pine were available and became common after 140,000 hectares (Figure B-2).

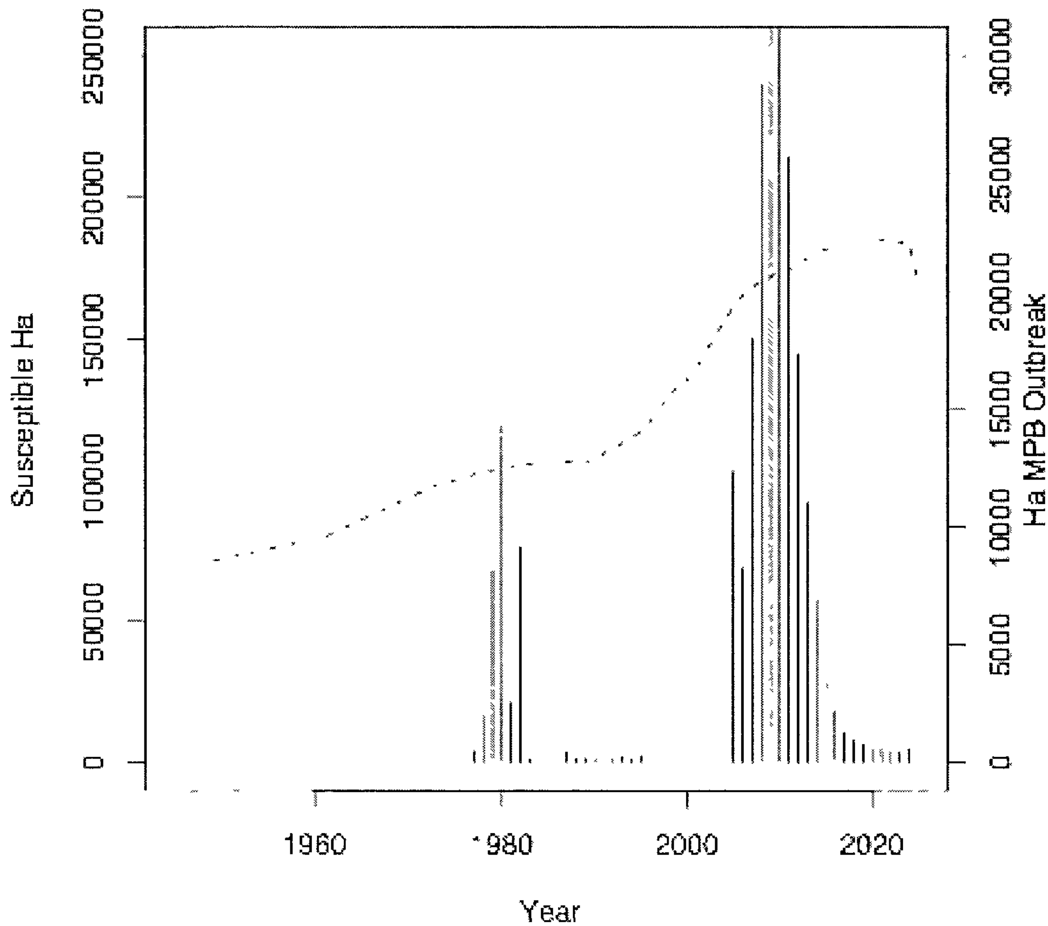

Figure B-2. Plot of historic and modelled susceptible pine from 1949 to 2025 (dashed line) and historic and downscaled provincial projection of area of MPB outbreak for the Cranbrook study area.

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Appendix C. East Kooteny (EK) ecosystem groups, including Biogeoclimatic Zone, Variant/Site Series and group name (Wells et al. 2004).

EK group #	Zone	Variant/Site Series	Name
1	PP	dh2-01	
1	IDF	dm2-03	subxeric-submesic IDF/PP
1	*IDF*	*un-DJ*	
2	***IDF***	***un-DP***	**submesic-mesic IDFun**
3	ICH	mk1-03	
3	ICH	mk1-04	
3	*ICH*	*dm-FB*	
3	IDF	dm2-01	cirum-mesic IDF/ICH/MS
3	IDF	dm2-04	
3	*IDF*	*dm2A-LP*	
3	MS	dk-04	
4	***ICH***	***dw-XA(01a)***	
4	***ICH***	***dw-XB(01b)***	**circum-mesic ICHdw/dm**
4	***ICH***	***dm-FH***	
5	*IDF*	*un2-FH*	mesic IDFun2
6	**ICH**	**mk1-01**	**mesic ICHmk1**
7	MS	dk-01	
7	*MS*	*dk-XF*	mesic MS/IDFdm2a
7	*IDF*	*dm2A-SG*	
8	**PP**	**dh2-03**	**subhygric PPdh2**
9	*IDF*	*un2-SD*	subhygric IDFun2
10	**ICH**	**mk1-06**	**subhygric ICHmk1**
11	MS	dk-05	
11	ICH	mk1-05	subhygric MS/ICH
11	*IDF*	*dm2A-SS*	
12	**IDF**	**dm2-05**	**subhygric-hygric (fluvial riparian) IDF**
12	***IDF***	***un-SS***	
13	*ICH*	*dm-XA*	subhygric-hygric ICH
14	**PP**	**dh2-04**	**hygric (fluvial mid-bench) PPdh2**
15	IDF	dm2-07	hygric (fluvial mid-bench riparian) IDF
15	*IDF*	*dm2-XB*	
16	*IDF*	*un-CD*	hygric (fluvial mid-bench riparian) IDFun
17	ICH	mk1-07	hygric (fluvial mid-bench riparian) ICH
17	*ICH*	*dm-SD*	
18	**MS**	**dk-06**	**hygric (fluvial mid-bench riparian) MS**
18	***IDF***	***dm2A-SH***	
19	MS	dk-07	subhydric MS
19	*IDF*	*dm2A-SB*	
20	**ESSF**	**dk-02**	
20	**MS**	**dk-03**	**xeric ESSF/subxeric MS**
20	***IDF***	***dm2A-LJ***	
21	ESSF	dk-03	
21	ESSF	dk-04	subxeric ESSF
21	*ESSF*	*dm2-FG*	
22	**ESSF**	**dk-01**	
22	**ESSF**	**dk-05**	
22	***ESSF***	***dm2-FP***	**mesic-subhygric ESSF**
22	***ESSF***	***dm2-FV***	
22	***ESSF***	***dm2-XC***	
23	ESSF	dk-06	hygric (fluvial riparian) ESSF
23	*ESSF*	*dm2-FH*	
24	*ESSF*	*dm2-FS*	subhydric ESSFdm2

EK group #	Zone	Variant/Site Series	Name
25	ESSF	wm-02	xeric subxeric ESSFwm dm1
25	ESSF	dm1 DB	
26	ESSF	dm1 FB	subxeric submesic ESSFdm1
27	ESSF	wm 03	submesic ESSFwm
27	ESSF	dm1-FA	
28	ESSF	wm 01	mesic ESSFwm
28	ESSF	dm1 FF	
29	ESSF	wm 04	subhygric ESSFwm
30	ESSF	dm1 FH	hygric (fluvial riparian) ESSFdm1
31	ESSF	dku FG	
31	ESSF	dmu1 FG	dry upper ESSF (Bl)
32	ESSF	dmu2 WH	
32	ESSF	wmu WH	dry upper ESSF (Pa)
32	ESSF	dku PJ	
32	ESSF	dmu1 PJ	
33	ESSF	dmu2 FB	
33	ESSF	wmu FB	
33	ESSF	dku HG	mesic upper ESSF (Bl)
33	ESSF	dmu1 HG	
33	ESSF	dmu2 FR	
33	ESSF	wmu FR	
34	ESSF	dku LG	
34	ESSF	dmu1 LG	mesic upper ESSF (La)
34	ESSF	dku LH	
34	ESSF	dmu1 LH	
35	ESSF	dku FH	
35	ESSF	dmu1 FH	subhygric upper ESSF (SE Bl)
35	ESSF	wmu WE	
35	ESSF	dmu2 WE	

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