University of Alberta

The Effects of Salvage Logging on Timber Supply and Wildlife Habitat

by

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A thesis submitted to the Faculty of Graduate Studies and Research in partial fulfillment of the requirements for the degree of Master of Science

in

Forest Biology and Management

Department of Renewable Resources

Edmonton, Alberta

Spring 2004

Your file Votre référence
ISBN: 0-612-96473-6
Our file Notre référence
ISBN: 0-612-96473-6

Abstract

Fire is an important natural disturbance agent in the boreal forest of Alberta. The most conspicuous result after fires is the large number of standing dead and dying trees. Salvage logging is conducted to minimize timber losses caused by fires. However, burned trees perform vital functions in forested ecosystems, such as providing habitat for wildlife. In Alberta, there are few explicit policies on salvage logging. The objective of this study was to evaluate salvage logging thresholds and develop tradeoffs between timber supply and wildlife habitat areas under different salvage scenarios. A Monte Carlo simulation of forest fire and an optimization-based forest harvesting model were used to project annual allowable cuts, net present values, and habitat areas over 200 years. Results showed the probability distributions of projected outcomes, which could be used to help determine the appropriate salvage rate.

In order to simulate salvage logging spatially, a spatially explicit fire model is necessary. A simple spatial fire model was developed to simulate the spread of wildfire at large spatial and temporal extents by using a cellular automaton approach. The model presented the landscape as a hexagonal-based lattice and as a square-based lattice with a 3 ha resolution, and compared the performance of fire spread with the hexagonal and the square models. The spread probability of a fire was modified from the annual burn rate of each fuel type. Results showed that both hexagonal and square models could represent well fire size distributions and annual burned areas, but the hexagonal model simulated fire patterns and fire skips more naturally than the square model.

Acknowledgements

First of all, great thanks go to Dr. Glen W. Armstrong, my supervisor, for his guidance, inspiration, assistance, and encouragement throughout the entire course of this project. I am also grateful to the members of my supervisory committee Drs. Ellen Macdonald and Fiona Schmiegelow for their guidance and advice. I would also like to thank Dr. Peter Boxall for his suggestions and comments.

Thank you to Mr. Rick Pelletier for his assistance in the GIS component and solving computer problems during the whole period of my research. I also thank my colleagues Steffen Mueller and Richard Simpson, and SIS labmates for their encouragement and assistance. I greatly appreciate their support and friendship during my research.

I am grateful for the continual encouragement and support from my family. Thank you for your love and understanding.

I would like to acknowledge the Sustainable Forest Management Network and the University of Alberta for funding and support during my research, and Alberta-Pacific Forest Industries Inc. for supplying forest inventory data.

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Chapter 1.

Introduction

1.1 The context of salvage logging

Salvage logging refers to "removal of selected portions or fragments of trees, which were felled and left behind during previous logging operations, or were damaged by windstorms, fire or other natural disturbances" (Newsom and Beasley 2000). This thesis focuses on salvage logging of burned trees after forest fires. Forests frequently suffer losses of timber from wildfire. Wildfire is a significant factor affecting many forested ecosystems, and leaves dead and dying trees behind. Salvage logging is used as an emergency measure for harvesting of fire-damaged trees before they deteriorate or become unmarketable due to insect damage and diseases (Barker 1989). Forest companies consider that salvaging dead and dying trees may recover the timber losses caused by fires. However, Carter (1992) and Beschta et al. (1995) argued that the large quantity of dead and dying trees remaining after fires perform vital functions in forested ecosystems. Ecologists and environmental groups are concerned about forest health issues such as wildlife habitat, soil properties and erosion, water quality, watershed protection, and other values of burned forests or trees (Henly 1988, McIver and Starr 2000). It is clear that removing the dead and dying trees from the forest will affect these functions. The additional disturbances of salvage logging to burned areas could alter post-fire colonization patterns (Crites and Hanus 2001), and could increase fire risk by leaving flammable branches, needles, and twigs on the forest floor afterwards (American Lands Alliance 2003). In addition, salvage logging reduces the forest canopy cover, allowing more sunlight to reach the forest floor, and consequently creating hotter and drier conditions (Kurulok 2004). Salvage logging is a controversial forest management activity (Anonymous 2000).

Fire effects on wildlife are complex because they are often indirect, affecting habitat more than individuals. Habitat conditions created by fires are biologically unique. Dead trees remaining after fires provide special habitat for some wildlife. Black-backed Woodpeckers (*Picoides arcticus),* for example, are considered dependent on newly burned forests (Saab and Dudley 1998, Hoyt and Hannon 2002, Stambaugh 2003, Stepnisky 2003), and Three-toed Woodpeckers (*Picoides tridactylus)* show higher abundance in burned and unsalvaged sites than in other habitat types (Hoyt and Hannon 2002, Saab and Dudley 1998, Greenberg et al. 1995, Hutto 1995, Stambaugh 2003, Stepnisky 2003). Such habitat may be difficult to create by logging, even with systems that leave standing residual trees, and therefore, burned habitat should perhaps be maintained on the landscape in some quantity. Questions have arisen whether or not salvage logging disrupts the ecological functions of burned stands and impacts forest health and productivity adversely over a long term. Although there are a number of studies concerning the effects of fires on either timber supply or wildlife habitat, there is almost no information concerning how salvage logging influences both timber supply and wildlife habitat.

In Alberta, there are few explicit policies related to salvage logging, and no criteria exist as to how much burned area should be left unsalvaged. Although the draft guidelines of the Fire Salvage Strategy Framework (SRD 2002) address some of the issues associated with salvage logging practices and incorporates more ecologically based planning, the emphasis is still on ensuring the volumes of green and burned timber to be harvested. The recommendations for maintaining the ecological functions of burned stands are vague. The Ontario Ministry of Natural Resources (OMNR 2001) published a guide to direct forest harvesting and salvage logging. It provided more details on cutting shape, spacing, and residual standards in both cutblocks and salvaging blocks. It suggested that salvage logging be included in forest management policy, and that the amount of burned forest being salvaged be decreased.

The fire and harvest residual (FaHR) project conducted by the Alberta Research Council (Lee 1999) compared the ecological characteristics of postfire and post-harvest stands over 28 successive years. This project considered to the management of postfire stands in which many special and unique biodiversity features are not replicated within conventional harvest blocks. Recent studies by Stambaugh (2003), Stepnisky (2003), and Kurulok (2004) further supported the ecological functions of burned stands. These studies suggested that a proportion of mature or over-mature burned stands should be left unsalvaged.

Hunter (1993) described the natural disturbance model (NDM) of forest management. He proposed that forest practices which emulate natural disturbances in terms of frequency, size and pattern, and residual material remaining can be developed. The implicit hypothesis of NDM is that biodiversity can be maintained if harvest practices approximate natural disturbances by creating harvest openings similar to what would have been created by nature. However, current forest policies in Alberta, such as lower timber dues and non-chargeability of salvaged volume against annual allowable cut (AAC) (AEP 2001), provide the motivation for timber companies to remove fire-killed and fire-damaged timber. Since fire-killed trees within forest management agreement (FMA) areas are considered an economic loss, it is common for forest companies to employ salvage logging operations following a fire in the forest stand. Salvage logging commences almost immediately after burning to avoid a decrease in timber value due to cracks or insect damage. The paradox is that salvage policies and practices encourage the creation of burned stands that are similar to clearcuts; at the same time with NDM management, cutblocks attempt to mimic burned stands. The examination of salvage policies and exploration of alternative strategies aim, therefore, to better understand the long-term impacts of the further disturbance in wildfire stands specifically caused by salvage logging.

In Chapter 2, I simulate different salvage logging alternatives and silvicultural scenarios, and describe their effects on timber supply and habitat areas for woodpeckers. I propose alternative salvage logging thresholds as the means to maintain burned habitat areas, while also meeting AAC requirements. Given the different rates of salvage logging, it is necessary to examine the differences in AACs over the planning period, while burned habitat areas are maintained. The interactions between a stochastic fire regime and salvage logging practices could have substantial implications for wildfire and forest management in sustainable forest development.

1.2 The context of spatial fire models

Fire disturbance models have been considered one of the important tools in the simulation of vegetation succession in landscape ecology (Mladenoff and Baker 1999). With the increasing interest in emulating natural fire disturbance patterns in forest management planning, an understanding of natural fire regimes is becoming more important. Natural fire disturbance is probabilistic at various spatial and temporal extents rather than deterministic (OMNR 2001). The extents range from hundreds and thousands to a few hectares in forest regions, and from hundreds of years to only minutes in fire effects over time. One of the challenges of simulating fire effects is the great range of spatial and temporal extents that must be considered to reflect the variations in characteristics of both fires and landscape structures (Gardner et al. 1999). One approach to emulating stochastic fire events and understanding fire dynamics is to use fire simulation models to analyze and predict fires. Such model simulations can provide a means to analyze and explore effects that cannot be observed and obtained directly, especially at large spatial and temporal extents (Reinhardt et al. 2001). Current fire models emphasize the fine-grained dynamics of fire behavior and on the identification of fire hazards, while the long-term interactions of landscapes with fire regimes have been ignored (Gardner et al. 1999), although this is changing (e.g. Cumming et al. (1998) and He and Mladenoff (1999) simulated long-term fire effect).

Fire disturbances have been simulated in different ways using methodologies suitable to the objectives and goals. There have been many efforts to simulate fire spread patterns on the basis of mechanistic principles which require detailed input information on weather conditions, topography, and the characteristics of fuel beds. Most such models, based on deterministic equations, are used to understand fire behavior and to explain how, why, and when a fire pattern forms based on fundamental fire principles at small extents (Finney 1999). They are particularly useful for investigating or reconstructing the causes for or pattern within a single fire (Reinhardt et al. 2001). In contrast to mechanistic models, probabilistic models are based on the empirical probabilities of fire spread and attempt to simulate only final burned patterns at broad extents. These models are designed to explore the potential results of various scenarios, in order to inform forest management decisions (Gardner et al. 1999, Hargrove et al. 2000).

Spatial simulation plays a critical role in explaining a wide range of ecological outcomes, such as the landscape dynamics after disturbances, or the distribution and location of fires over time. One widely used method for simulating fire spread is a cellular automata (CA) approach combined with geographic information system (GIS) techniques. The CA approach is a dynamic system in which space and time are discrete. A CA model consists of a regular grid of cells that interact with their neighbors. By receiving input from connected cells, a cell uses rules to determine what its reaction should be (Wolfram 1984). The rules can be deterministic or probabilistic. CA has become an important mechanism for investigating pattern formation, such as forest fires. Spatial heterogeneity has been shown to affect the propagation of disturbance through a landscape (Von Niessen and Blumen 1988, Hargrove et al. 2000, Berjak and Hearne 2002). A variety of spatial fire models have been used to establish relationships between fires and the features of the environment such as climate, topography, and fuel conditions. However, most of them have set out to analyze and predict the dynamics of fire behavior at a small extent so as to reduce fire hazard and risks (Von Niessen and Blumen 1988, Perry 1998, Gardner et al. 1999). Such models are designed to analyze the spread of a single fire rather than a fire regime (Baker 1999). Few models are available to simulate fire regimes or to project fire size distributions, fire patterns, and fire islands at larger spatial and temporal extents.

In Chapter 3, I develop a spatial fire model. I present a CA model for simulating fire spread that considers major factors that influence fire propagation, such as fuel type, wind speed and direction. The model uses probabilistic functions to simulate fire size distributions, fire patterns (shape, skips), and annual burned areas in the northeastern Alberta over a 40-year horizon. The landscape is presented as a hexagonal-based lattice and a square-based lattice with a 3 ha resolution each. The simulations of fire spread with the hexagonal and square fire models are compared. Advantages that the hexagonal model enjoy over the square model include the equal distance of any cell to any of their neighboring cell, a closely packed structure, a better approximation of a circle, and fewer variations in probability of cells' catching fire from neighboring burning ones.

In the final chapter, I summarize the study's results, present their potential implications for forest management practices, and provide suggestions for future research.

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Chapter 2.

The effects of salvage logging on timber supply and wildlife habitat

2.1 Introduction

Increasingly, forest management policies are encouraging ecologically sound forest management practices, and forestry companies are attempting to emulate natural disturbances across the landscape to maintain biodiversity (Hunter 1993, Hutto 1995, Stelfox 1995, OMNR 2001). This forest policy realignment is leading away from clearcutting, and towards practices thought to benefit biodiversity in forest ecosystems. According to the natural disturbance model (Hunter 1993), if logging practices emulate wildfires as closely as possible, biodiversity may be maintained in harvested stands in the same way that it is in postfire stands over the long term. However, studies have found that in postfire and post-harvest areas characteristics of plant communities such as species compositions, wildlife communities, soil nutrient flow, etc., remain different for as long as 28 years following disturbances (Lee 1999).

The greatest difference between postfire and post-harvest landscapes comes from the large number of standing dead and dying trees that immediately appear in postfire forests and their decline thereafter (Lee 1999, Song 2002), providing wildlife communities in the interim with areas for foraging, nesting, sheltering, and perching. Stands affected by burning are desirable for some species because of the unique biotic characteristics of burned forests (Schieck and Hobson 2000). Although the wildfire template is used in forest harvest practice, harvest operations are unlikely to produce stands that function similarly to wildfire origin stands. From their physical configuration to their biotic characteristics, burned stands present many differences from harvested stands, while fire-created snags and downed logs perform many unique and vital ecological functions for forest soils, watersheds, vegetation, and wildlife (Stelfox 1995, Blank and Zamudio 1998, Lee 1999, McIver and Starr 2000, Schieck and Hobson 2000, Song 2002, Stambaugh 2003, Stepnisky 2003, Kurulok 2004). Further, the natural fire regime is highly variable, both spatially and temporally, so it is difficult to make forest harvesting mimic natural fires (Schneider 2002).

In the boreal forest landscapes of Alberta, wildfire is a significant disturbance (Lee 1999, Song

2002), and management practices must deal with the challenges related to forests affected by fires and subsequent salvage logging operations (Dietz 1997, Dietz 2000). Fire-damaged trees within Forest Management Agreement (FMA) areas are considered a financial loss if they are not salvaged. Alberta forest legislation and regulations allow, and even encourage, forest companies to harvest fire-killed timber. For example, if forest tenure holders do not salvage fire-damaged merchantable trees, the volume of unsalvaged timber may be charged against their annual allowable cut (AAC) or quota (AEP 2001). Merchantable dead wood is usually not included in AAC determination and is charged at lower timber dues (AEP 1998, Alberta Government Services 2003). It is very common in FMA areas that merchantable fire-damaged trees are salvaged immediately after burning, before the wood quality declines due to cracking, insect damage, diseases, or blow down.

The Fire Salvage Strategy Framework (LFD 2002) provides a framework for the management of salvage logging in Alberta. This Framework defines salvage logging objectives in terms of salvage allocations, annual allowable cut recalculations, reforestation strategies, and production control. Although it incorporates certain ecologically based planning into salvage logging practice – it for example dictates leaving a proportion of snags – the priorities are still to ensure that fire-damaged wood contributes to maintaining timber supply and accrues as a financial benefit to the forest industry. Salvage logging operations are normally carried out by forest companies in forest management units (FMUs) (Crites and Hanus 2001). Lower timber dues for fire-damaged timber provides an incentive for forest companies to take advantage of this wood to increase their annual revenues. In addition, under Alberta's forest tenure system, timber quota holders are allocated a specific proportion of AAC in FMUs; salvage logging of burned trees may provide them opportunities to compensate for timber loss caused by fires. Salvage logging is looked upon by the timber industry as an alternative way to meet timber demands and to generate revenues.

Nevertheless, there are few explicit prescriptions available to guide salvage practices, or to determine the percentage of merchantable burned stands to be protected from salvage logging, or to define where and how this protection should be done. Since wildfire is a highly variable natural disturbance (Armstrong 1999), it is difficult to set down a logging template to direct salvage operations. The criteria for determining what kinds of trees and how much burned timber should be removed still remain unclear.

The objective of this study was to evaluate the appropriate rates of salvage logging under natural fire regimes. In this model, alternative salvage logging thresholds were proposed as the means to maintain burned habitat areas, while meeting the AAC requirements. With the rates of salvage logging determined, it is necessary to examine the differences in AACs affected over the planning period, while certain burned habitat areas are maintained. Doing so may provide insights into the importance of balancing the harvesting of fire-damaged timber as a way for companies to recover losses from fire and examination of the management of burned land. Alternative thresholds may provide information about recovering timber while meeting environmental protection or other needs in specific forest management areas.

The influences of fire on timber supply have been extensively studied and modeled (Van Wagner 1983, Reed and Errico 1986, Reed 1994, Boychuk and Martell 1996, Armstrong 2000), but most studies have ignored the effects of salvage logging because they assumed that the burned stands were regenerated immediately after fires. Cumming et al. (1994) and Armstrong et al. (2003) ignored the burned forest as special habitat when they simulated the potential conflicts between timber supply and wildlife habitat following fire disturbances. Understanding the effects of salvage logging practices on both burned habitat and timber supply in the long term, and on net present value (NPV) of harvested volumes over the planning period, is necessary if forest managers hope to maximize the long-term objectives of the timber industry while at the same time maintaining areas of special habitat for wildlife.

2.2 Study Area and Input Data

The study area, which includes eight FMUs, is located in the southern part of the FMA area of Alberta-Pacific Forest Industries Inc. (Figure 2.1). The Alberta-Pacific FMA area is largely boreal mixedwood forest, dominated by aspen (*Populus tremuloides*) and white spruce (*Picea glauca*), the prevalent sources of pulp and softwood timber products. Jack pine (*Pinus banksiana*) and black spruce (*Picea mariana*) are widely distributed but seldom develop to commercial stands. Tamarack (*Larix laricina*) is a noncommercial species that is not harvested in any substantial quantity. In this region, there is one large pulp mill owned by Alberta-Pacific, and several sawmills of varying sizes. Inventory data in the study area track recent forest conditions over a landscape area of 2,356,044 ha; this total area is further divided into a productive, or net merchantable landbase, and a passive landbase. The productive landbase is used for timber harvesting activities, while the passive landbase is considered unharvestable area for riparian buffer zones, or as a site for subjective deletions and other operational considerations. On the basis of the Alberta Vegetation Inventory (AVI), stands with predominantly deciduous species (i.e. where 80% or more of the canopy is deciduous) are classified as pure deciduous stands; these include pure aspen, pure balsam poplar, pure birch, and mixed deciduous species. Stands in which more than 20% and fewer than 80% of the species are coniferous are classified as mixedwood stands. The others are classified as pure coniferous stands (i.e. as white spruce, black spruce, and pine stands).

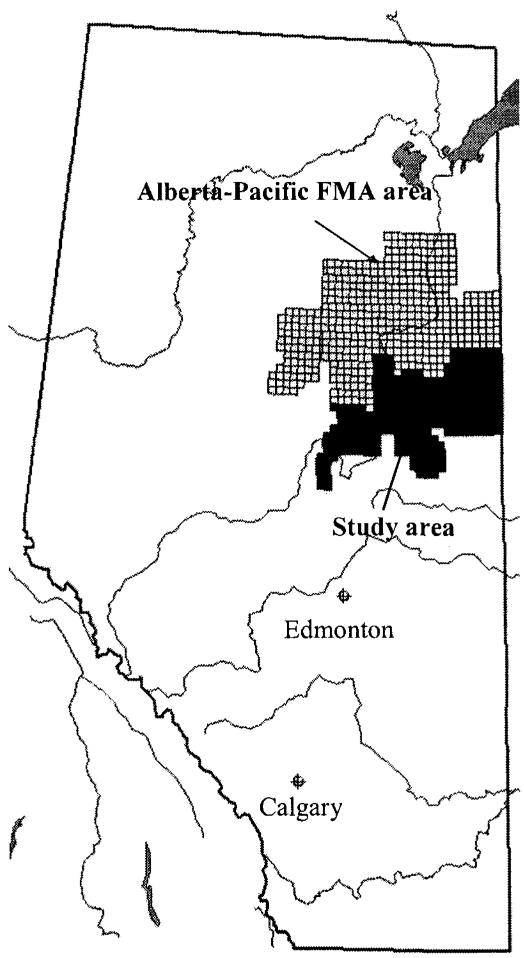


Figure 2.1. Location of study area, Alberta-Pacific Forest Industries Inc., Alberta

Harvest operations in this study closely follow the practices set out in the Alberta-Pacific Operating Ground Rules (Alberta-Pacific Forest Industries Inc. no date), which exclude harvesting activities in certain areas. These include the buffer zones of watercourses and areas adjacent to water and stream features, non-forest areas, and areas considered unsuitable for harvesting operations. The areas deemed unsuitable include stands containing non-commercial species, non-commercial coniferous and deciduous stand densities, non-commercial site index (height-age relationship), or non-commercial timber productivity rating (timber productivity rating (TPR) is "U"). Because only the productive landbase meets the requirements for timber harvesting, it is included in the harvest model.

Table 2.1 provides a summary of landbase classification for the study area: 40.29% of the total area is composed of productive stands containing merchantable volumes of commercial tree species; and 59.71% is composed of non-forest areas, unproductive areas, and non-commercial species areas. The initial forest age structures for each of the cover types in the productive and the passive landbases are shown in Tables 2.2 and 2.3.

Table 2.1. Summary of landbase classification for the study area.

Landbase	Cover type	Area (ha)	%
Productive	Deciduous	436,103	18.51
	Deciduous-dominated mixedwood	62,906	2.67
	Conifer-dominated mixedwood	41,466	1.76
	Conifer	408,773	17.35
	Total	**949,250**	**40.29**
Passive	Buffer/preserved areas	57,958	2.46
	Unproductive areas	283,903	12.05
	Non-merchantable stands	965,035	40.96
	water	99,896	4.24
	Total	**1,406,793**	**59.71**
Total		**2,356,044**	**100**

The growth and yield tables used in this study are Alberta Phase 3 inventory yield tables (Alberta Forest Service 1985). Table 2.4 provides the net volume (m^3/ha) and the quadratic mean diameter (qdbh) (cm) of a stand in the productive landbase by stand age, cover type and TPR at the 15/10+ cm utilization standard. Merchantable harvestable stands refer to stands with a net volume greater than 50 m^3/ha at the 15/10+ cm utilization standard. Since the yield table in unproductive areas (TPR is U) is not available, and the model attempted to project burned habitat areas in the unproductive landbase as well, I estimated the net volume in the unproductive stands based on the Alberta Phase 3 inventory yield tables in productive landbase (Alberta Forest Service 1985).

Table 2.2. Initial forest inventory (area in ha) in the productive landbase by stand age, cover type and timber productivity rating (TPR). G, M, and F represent the good, median, and fair TPR sites respectively.

Age	Deciduous			Mixedwood			Pine			Black spruce			White spruce		
(years)	G	M	F	G	M	F	G	M	F	G	M	F	G	M	F
10	1,480	278	0	70	183	0	605	4,151	362	0	6	0	250	200	0
20	8,466	2,979	43	992	1,435	38	5,290	10,778	3,571	79	296	96	806	1,716	0
30	2,854	3,214	3	941	578	0	2,552	290	0	22	0	0	35	17	0
40	8,149	15,685	464	549	1,496	109	560	1,840	2	255	24	0	285	73	5
50	9,883	5,060	640	993	1,519	197	2,528	5,547	196	1,479	42	0	398	128	6
60	53,154	10,691	1,377	4,003	3,073	609	18,402	23,588	1,116	6,614	264	0	652	353	36
70	65,811	11,687	535	4,146	2,800	245	12,057	27,438	1,214	9,730	12,276	8	1,404	322	7
80	77,355	7,324	297	5,540	2,042	160	5,426	14,357	1,543	12,948	9,194	14	3,322	1,120	5
90	28,975	4,309	69	3,934	1,487	93	1,945	9,359	530	7,753	20,615	59	3,233	1,449	4
100	32,879	4,447	212	7,208	2,330	155	1,966	6,439	585	13,992	34,782	0	4,808	3,566	10
110	18,703	4,605	62	7,559	1,836	24	1,724	4,799	830	6,378	9,689	40	5,267	2,395	31
120	13,125	4,502	145	9,328	3,034	30	361	2,742	365	4,361	12,646	7	7,262	4,044	15
130	4,586	1,269	15	2,636	2,300	8	115	886	124	1,361	5,542	0	2,230	4,027	20
140	4,428	5,368	182	7,226	5,974	102	261	1,072	115	1,999	5,346	2	8,888	6,842	114
150	227	505	37	638	2,711	15	58	294	119	516	1,964	187	1,231	3,459	21
160	61	252	8	386	1,642	17	13	209	76	168	446	26	521	3,544	17
170	0	25	0	116	135	10	0	23	20	99	150	25	85	96	0
180	1	0	20	5	43	18	0	26	0	10	2	10	26	153	2
190	4	0	7	0	10	0	0	11	23	0	115	34	3	7	0
200	3	0	0	0	0	0	0	0	2	0	15	18	15	23	0
Total	330,143	82,202	4,115	56,268	34,628	1,832	53,862	113,847	10,795	67,765	113,415	526	40,722	33,532	293

Table 2.3. Initial forest inventory (area in ha) in the passive landbase by stand age, cover type, and timber productivity rating (TPR). G, M, F, and U represent the good, median, fair, and unproductive TPR sites respectively.

Age	Deciduous				Mixedwood				Pine				Black spruce				White spruce			
(years)	G	M	F	U	G	M	F	U	G	M	F	U	G	M	F	U	G	M	F	U
10	26	1	0	0	10	4	0	0	2	161	6	0	0	2	0	0	0	0	0	0
20	1,629	1,344	28	0	70	1,951	80	0	1,090	16,741	10,255	0	1,778	7,617	4,559	93	2	18	0	0
30	249	857	10	0	69	753	76	0	150	1,613	3,022	2	2,726	9,139	8,934	827	4	36	0	0
40	673	1,607	563	0	235	405	460	0	41	77	476	6	11,285	158	11,046	2,110	22	120	5	0
50	793	1,011	346	82	180	564	401	23	317	1,325	243	1	8,332	22,147	22,411	7,334	344	203	16	0
60	2,997	1,080	359	185	524	968	629	81	3,403	5,958	392	23	8,575	74,315	45,657	22,919	214	201	19	0
70	3,740	954	136	87	412	909	225	86	1,544	9,250	768	29	6,947	98,440	28,853	23,675	527	361	32	0
80	4,816	643	77	79	623	793	143	34	267	3,721	564	15	6,893	70,064	20,850	14,885	525	493	46	1
90	1,936	485	21	42	254	420	143	76	121	1,636	445	1	1,518	18,723	23,153	10,281	430	401	12	0
100	2,649	388	24	60	677	359	202	60	147	986	346	38	2,220	20,609	20,319	15,130	687	762	30	18
110	1,361	460	6	0	540	229	85	47	127	278	165	0	957	5,869	5,940	5,507	842	505	3	0
120	1,062	374	7	8	695	331	33	25	17	144	268	3	825	5,482	1,851	2,633	1,143	747	2	3
130	324	63	0	4	198	264	12	8	3	47	48	5	173	2,034	809	1,367	308	693	29	9
140	293	468	18	1	668	493	43	39	6	47	12	0	293	2,446	857	1,765	1,375	785	33	0
150	56	18	0	0	43	157	7	0	0	19	11	0	46	467	215	561	174	401	19	0
160	11	38	0	3	50	115	5	10	0	17	0	0	8	90	41	200	107	523	1	0
170	0	3	0	0	5	3	2	0	0	0	0	0	5	17	2	183	11	3	0	0
180	2	0	0	0	0	1	0	2	0	0	0	0	0	4	11	5	1	8	2	0
190	0	0	8	0	0	2	0	0	0	0	1	0	6	13	3	86	0	0	0	0
200	27	0	0	0	0	0	0	0	0	0	0	0	0	33	41	228	5	5	0	0
Total	22,643	9,794	1,604	552	5,251	8,721	2,546	489	7,235	42,021	17,023	124	52,586	337,670	195,553	109,789	6,721	6,264	249	31

Table 2.4. Volume yield (m^3/ha) and quadratic mean diameter (qdbh) (cm) by stand age, cover type, and timber productivity rating (TPR) at the utilization standard 15/10+ cm. G, M, and F represent the good, median, and fair TPR sites respectively.

	Deciduous						Mixedwood						Pine						Black spruce						White spruce					
TPR	G		M		F		G		M		F		G		M		F		G		M		F		G		M		F	
Age	qdbh	vol	qdbh	vol	qdbh	vol	qdbh	vol	qdbh	vol	qdbh	vol	qdbh	vol	qdbh	vol	qdbh	vol	qdbh	vol	qdbh	vol	qdbh	vol	qdbh	vol	qdbh	vol	qdbh	vol
10	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
20	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
30	14	10	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
40	18	98	14	8	0	0	17	20	0	0	0	0	16	56	0	0	0	0	0	0	0	0	0	0	18	22	0	0	0	0
50	22	164	17	68	0	0	21	60	15	9	0	0	18	131	16	33	0	0	16	6	0	0	0	0	20	91	17	2	0	0
60	25	215	19	115	15	10	24	95	19	37	0	0	19	199	17	83	0	0	16	56	0	0	0	0	22	156	19	50	0	0
70	27	255	21	154	16	44	27	121	22	63	16	13	21	257	18	129	16	17	17	105	0	0	0	0	24	212	20	96	0	0
80	30	284	23	184	17	72	28	141	24	87	18	32	23	308	18	170	16	46	18	150	0	0	0	0	26	262	22	140	17	13
90	32	310	24	210	18	96	29	154	25	108	20	51	25	350	19	207	17	73	18	192	16	25	0	0	28	305	23	179	18	43
100	34	331	26	231	19	116	29	163	27	124	22	68	26	385	20	239	17	98	19	231	16	51	0	0	29	341	24	215	19	71
110	35	267	27	186	20	93	30	140	28	106	23	59	28	415	21	268	17	121	20	267	17	77	0	0	30	371	25	247	20	98
120	37	204	28	142	21	71	30	116	28	89	25	49	29	441	22	293	18	142	21	299	17	102	0	0	31	397	26	276	21	124
130	38	140	29	98	21	49	30	93	29	71	26	39	30	464	23	316	18	162	21	328	17	127	0	0	32	420	27	301	22	149
140	39	76	30	53	22	27	30	70	29	53	26	29	31	484	24	336	19	180	22	354	18	150	16	10	33	438	28	324	23	172
150	40	13	31	9	23	4	30	47	30	35	27	20	32	501	25	354	19	195	23	379	18	173	16	25	34	455	29	343	23	194
160	41	0	31	0	23	0	30	23	30	18	28	10	33	516	26	370	20	211	23	401	19	195	16	40	35	479	30	362	24	213
170	41	0	32	0	24	0	30	0	30	0	28	0	34	530	26	385	20	223	24	422	19	216	16	55	35	483	31	378	25	233
180	42	0	33	0	24	0	30	0	30	0	29	0	35	542	27	398	20	236	24	440	19	236	17	69	36	494	31	393	26	250
190	42	0	33	0	24	0	30	0	30	0	29	0	35	542	27	398	20	236	24	440	19	236	17	69	36	494	31	393	26	250
200	42	0	33	0	24	0	30	0	30	0	29	0	35	542	27	398	20	236	24	440	19	236	17	69	36	494	31	393	26	250

2.3 Model Descriptions

2.3.1 Fire model

Fire is an inherent risk that has the potential to adversely affect timber supply and fire management. A number of studies have demonstrated that the long-term timber supply and the economic benefit from a forest can be reduced by the presence of fire. This finding applied whether the study used a constant burn rate (Van Wagner 1978, Reed and Errico 1986), two-point discrete constant burn rate (Boychuk and Martell 1996), or continuous stochastic burn rate with a lognormal distribution (Armstrong 1999). Boychuk and Martell (1996) argued that wildfire is highly variable and should be simulated using a stochastic process. Using a constant burn rate might be inappropriate in stochastic forest ecosystems (Armstrong 1999, OMNR 2001).

Armstrong (1999) used statistical methods to calculate and analyze the distribution of fire burn rates. He formulated the annual burn rate in the boreal forest as a lognormal distribution. The results suggested that the annual burn rate in the boreal mixedwood forest is highly variable, and it did not result in an equilibrium age-class forest structure in his study area, as Van Wagner thought it would (1978). This lognormal fire regime generates a continuous representation of the annual burned area, and it better reveals the stochastic features of fires. The probability density function of the lognormal distribution is

$$f(x) = (\sqrt{2\pi}\sigma x)^{-1} \exp\left[-\frac{(\ln x - \mu)^2}{2\sigma^2}\right]$$

where

x = the annual proportion of the area burned
μ = the mean of the natural logarithm of x
σ = the standard deviation of the natural logarithm of x

Mueller (pers. comm., University of Alberta, April 2003) further explored the lognormal fire regime based on the study by Armstrong (1999) and identified the parameters μ and σ as -9.144 and 2.774, given an area of approximately 2.25 million ha, which is similar to my study area. Thus, in the present study's model, Armstrong's lognormal fire regime and Mueller's parameters were incorporated. The burn rate was constrained to be 0.20 or less in order to prevent the burned area in a year much larger than the historical fire record. In the model, the fire burn rate λ in year t was randomly drawn from the following distribution:

$$\lambda_t = \min(0.20, \exp x_t), \quad x_t \sim N(\mu, \sigma^2), \quad t = 1, 2, \ldots\ldots, 200.$$

2.3.2 Ecological effects

Timber supply and habitat areas for woodpeckers are the main criteria in this study for evaluating the sustainability of forest management. Fire disturbance creates great differences in the forest structure; biologically unique areas are created from the biomass of snags and downed woody materials, and these are absent from regular harvested areas (Lee 1999). Immediately following a fire, dense areas of snags dominate (Schieck and Hobson 2000). Large snags play an essential role in providing unique and special habitat for wildlife. Dead and dying burned trees, standing or downed, supply essential nutrients to the soil for new vegetation (Lee 1999, Song 2002). Numerous studies have demonstrated the importance of burned trees to some birds (Hutto 1995, Greenberg et al. 1995, Saab and Dudley 1998, Hobson and Schieck 1999, Schieck and Hobson 2000, Morissette et al. 2002, Stambaugh 2003, Stepnisky 2003). Snags fall down over time, and these downed woody materials provide the nesting materials, as well as foraging habitat, for some vertebrates (Song 2002) and many invertebrates and plants. Such snags originated from fires have high ecological value if they are left in place. The aim of salvage logging is to recover the fire-damaged timber, but this has a negative impact on some of the wildlife species.

Many studies have shown that recently burned areas contain fire-dependent species and irreplaceable habitat characteristics; these both contribute to the richness and the abundance of wildlife, in particular to some bird communities (Hutto 1995, Saab and Dudley 1998, Hobson and Schieck 1999, Hoyt 2000, Hoyt and Hannon 2002, Morissette et al. 2002, Stambaugh 2003, Stepnisky 2003). The reasons that certain bird species proliferate in recently burned forests can be credited to the increase, after fires, in the number of bark beetles and wood-boring larvae, and, because cones open in response to fire's heat, to the availability of conifer seeds that results (Hutto 1995). Three-toed Woodpeckers (*Picoides tridactylus*) and Black-backed Woodpeckers (*Picoides arcticus*) are particularly associated with recently burned forests in Alberta, because of the large numbers of dead and dying trees that become available (Saab and Dudley 1998, Hobson and Schieck 1999, Hoyt and Hannon 2002). In this study, Black-backed and Three-toed Woodpeckers were chosen as indicators for reflecting the area of habitat types under different salvage logging strategies. Habitat in this study is an area defined by its cover type and the time since its burning, i.e. its postfire time period.

Black-backed Woodpecker (BBWP)

Black-backed Woodpeckers appear to be associated primarily with burned areas and are deemed a burn-dependent species (Hoyt and Hannon 2002). They drill or excavate in sapwood for wood-boring beetle larvae. Murphy and Lehnhauser (1998) showed that Black-backed Woodpeckers, within three years of a fire, exploit the outbreaks of wood-boring beetles in dead and dying conifers (especially spruce species). However, Hoyt and Hannon (2002) detected Black-backed Woodpeckers' occupying burned stands for as many as eight years after a fire, owing to the presence of jack pine in their study area. Jack pine, with its thick bark, can retain moisture longer and thus has a higher capacity to survive fire. The prolonged occupancy of Black-backed Woodpeckers was due to the prolonged availability of beetle larvae for foraging (Hoyt and Hannon 2002).

Three-toed Woodpecker (TTWP)

Three-toed Woodpeckers occur in old coniferous or conifer-dominated mixedwood stands (Godfrey 1986), as well as in newly burned stands (Hoyt 2000). They nest in snags or partially dead conifers, preferably white spruce, black spruce or fir, and occasionally in deciduous trees. They mainly prey on bark beetle larvae, foraging by scaling bark off dead or dying conifers (Godfrey 1986). Hoyt and Hannon (2002) found that Three-toed and Black-backed Woodpeckers have different periods of occupancy, depending on the elapsed time postfire. Three-toed Woodpeckers are abundant only in newly burned areas less than three years postfire; afterwards, they move to unburned old coniferous forests, because of the decreased numbers of bark beetle in burned areas.

Both woodpecker species require large diameter conifers for nesting and foraging. Studies provide significant evidence of their nesting and feeding on big snags, rather than on smaller trees (Hutto 1995, Saab and Dudley 1998, Hoyt and Hannon 2002). Forest companies are likewise only interested in merchantable fire-killed trees, as these are the most usable for recovering fire losses and gaining revenues. In this model, I assume that only burned stands with qdbh greater than 15 cm are subject to both salvage logging and providing suitable wildlife habitat. This is a compromise figure; other studies have chosen dbh 10 cm (Hutto 1995), 12 cm (Stepnisky 2003), 14.7 cm (Hoyt and Hannon 2002), and 25 cm (Saab and Dudley 1998) as the levels of providing suitable habitat for woodpeckers. Burned stands with dbh less than 15 cm are not considered in the habitat model.

Despite the fact that there are great differences in characteristics between harvested stands and burned stands, there is evidence that some important characteristics of these stands gradually converge over 28 years postfire for boreal mixedwood forests (Lee 1999). The rate of snags falling in the first two years postfire is very low. Even within ten years of a fire, approximately 25% of snags will have fallen. But, after ten years, the rate of falling increases greatly (Lee 1998). On the basis of dynamics of snags (Lee 1998), the postfire habitat was categorized into three stages in this study (Table 2.5). Stage 1 comprises the newly burned areas with large numbers of snags, and is the stage which provides habitat for woodpeckers or merchantable snags. Stage 2 presents fewer snags than Stage 1, with some already having fallen, and serving as habitat. More downed woody materials appear on the forest floor in this stage. In stage 3, the snag density rapidly decreases, and large quantities of downed woody materials appear on the forest floor to serve as shelter and food sources for some vertebrates. In the meantime, rotted woody material creates a cycling nutrient pool. More and more new vegetation appears in the burned stands. This stage presents special ecosystem characteristics. In stages 2 and 3, no salvage logging occurs because snags in these periods will have become subject to insect attacks, cracks, and decaying, all of which make them undesirable for lumber or pulp production.

Table 2.5. Postfire habitat stages

Stage	Years postfire
1	1 – 2 years
2	3 – 10 years
3	11 – 30 years

Table 2.6 shows the preferred habitat, described by cover type and postfire habitat stage, of the Black-backed and Three-toed Woodpeckers. The model presented below will present the changes in these birds' habitat areas under various salvage logging strategies.

Table 2.6. Black-backed and Three-toed Woodpeckers preferred habitat

Woodpecker	Cover type	Postfire habitat stage		
		1	2	3
Black-backed Woodpecker	White spruce	×		
	Black spruce	×		
	Pine	×	×	
	Mixedwood	×	×	
	Deciduous			
Three-toed Woodpecker	White spruce	×		
	Black spruce	×		
	Pine			
	Mixedwood	×		
	Deciduous			

2.3.3 Harvest model

In the study area, Alberta-Pacific has rights to harvest deciduous species, and a number of coniferous timber quota holders harvest coniferous timber. The AAC is defined as the maximum annual amount of timber that can be harvested on a sustainable basis within the FMA area. Current forest management legislation in Alberta requires that timber yield be sustained in perpetuity; the declared intent of the forest industry and the government is to "look at the possible impacts of today's harvesting practices on a forest 120-240 years in the future" (AEP 1994). AACs are determined based on maximum timber harvest levels after meeting harvesting constraints over two forest rotation lengths, i.e. for about 200 years.

Harvest planning in the merchantable forest stands of the productive landbase operates according to constraints. Forty percent of the study area contains productive merchantable stands that produce potentially harvestable volumes of commercial species. The objective function of the timber harvest model is to maximize the annual allowable harvest volumes of green timber over 200 years. The constraints, which provide the measures to assess the sustainability of timber production, are as followings:

- Even-flow AACs, for both deciduous and coniferous species over a 200-year planning horizon, are required.
- The growing stock is set to be non-declining in the last 100 years of the planning horizon, as the Alberta-Pacific runs in harvest planning.
- 5% of commercial timber is left for wildlife and another 5% for isolated, protective, or reserved areas. This was accomplished by reducing the yields of commercial timber by 10%. (see Appendix 1).

The harvest model attempted to achieve the goal without fire disturbance effects by assuming that only harvest operations occur. Harvested stands were assumed to regenerate to the same species without delay across the entire landbase. This model was constructed by using the Woodstock Forest Modeling System (Remsoft Inc. 2000), which is a flexible system for solving forest management problems and is capable of generating linear program matrices.

Conversion return

The conversion return is the difference between the selling price of wood products in the market less all costs connected to both getting the timber from stand to mill gate and to its further processing (Davis et al. 2001). Average selling prices in 2003 were 356.76 C$/Mfbm[①] for coniferous lumber, and 742.57 C$/ADMt[②] for bleached kraft pulp (SRD 2003). The average timber dues in Alberta were 3.65 C$/$m^3$ for green conifers and 0.42 C$/$m^3$ for deciduous trees (AEP 2001). The province's timber dues for fire-killed trees are at 0.70 C$/$m^3$ for conifers and at 0.42 C$/$m^3$ for deciduous in the Alberta-Pacific FMA area (AEP 2001). Burned timber was assumed to produce the same products as green timber. Tables 2.7 and 2.8 show the costs used to calculate the conversion return for coniferous and deciduous species. The total costs are approximately 272 C$/Mfbm for sawmills (Pearse 2001) and 685 C$/ADMt for pulp mills (pers. comm. Messmer, M. Weyerhaeuser Company, 2002). The processing costs for burned wood are assumed to be 5% higher than for green wood. General regeneration costs in regular harvested coniferous stands are approximately 920 C$/ha (Rodrigues 1998). The regeneration costs in burned stands are estimated to be 10% higher than those in harvested stands, i.e. approximately 1012 C$/ha, to reflect the efforts to remove logging debris and clear charcoal pollution. Aspen regenerates very quickly following disturbances (Fraser et al. 2003), so deciduous stands are normally left to regenerate naturally (Alberta-Pacific Forest Industries Inc. 1999). Therefore, the regeneration costs in the deciduous stands are assumed to be 0. The annual discount rate is assumed to be 5%. The net present value (NPV) is calculated using the following equations (Davis et al. 2001):

$$NPV_r = \sum_{n=1}^{200} \left(\frac{Rt_1 \times V_1}{(1+i)^n} - \frac{C_1 \times A_1}{(1+i)^n} \right)$$

$$NPV_b = \sum_{n=1}^{200} \left(\frac{Rt_2 \times V_2}{(1+i)^n} - \frac{C_2 \times A_2}{(1+i)^n} \right)$$

$$NPV = NPV_r + NPV_b$$

where:
NPV_r = net present value from regular harvested timber
NPV_b = net present value from salvaged timber
NPV = total net present value
Rt_1 = conversion return for green timber
Rt_2 = conversion return for burned timber
C_1 = regeneration cost in regular harvested stands
C_2 = regeneration cost in salvaged stands
V_1 = volume of green timber harvested
V_2 = volume of burned timber harvested
A_1 = area of regular harvested stands
A_2 = area of salvaged stands
i = interest rate 5%
n = planning horizon, 200 years

① Thousand board feet
② Air dried metric ton

Table 2.7. Conversion returns for coniferous timber

	Selling price	Timber dues	Sawmill cost	Conversion return **
Green wood	356.76 C$/Mfbm		272.00 C$/Mfbm	
	82.97 C$/m^3 *	3.65 C$/m^3	63.26 C$/m^3 *	16.06 C$/m^3
Burned wood	356.76 C$/Mfbm		285.60 C$/Mfbm	
	82.97 C$/m^3 *	0.70 C$/m^3	66.42C$/m^3 *	15.85 C$/m^3

* For 1 Mfbm of lumber approximately 4.3 m^3 of wood is required (Alberta Forest Service 1988).
** The conversion return is the selling price less timber dues, logging costs and manufacturing costs.

Table 2.8. Conversion returns for deciduous timber

	Selling price	Timber dues	Pulp mill cost	Conversion return **
Green wood	742.57 C$/ADMt		685.00 C$/ADMt	
	157.99 C$/m^3 *	0.42 C$/m^3	145.74 C$/m^3 *	11.83 C$/m^3
Burned wood	742.57 C$/ADMt		719.25 C$/ADMt	
	157.99 C$/m^3 *	0.42 C$/m^3	153.03 C$/m^3 *	4.54 C$/m^3

* For 1 ADMt of bleached kraft pulp approximately 4.7 m^3 of wood is required (Alberta Forest Service 1988).
** The conversion return is the selling price less timber dues, logging costs and manufacturing costs.

2.3.4 Salvage logging model:

Salvage logging normally occurs in a relatively short time frame after fires (i.e. less than 2 years postfire), since the dead trees substantially decrease in value. In this study, salvage logging was assumed to happen in the first year postfire (SRD 2002). After reviewing government policies and salvage logging guidelines, I identified the following salvage logging alternatives and stand treatment scenarios for salvaged and unsalvaged stands:

Alternative 1. Salvage nothing after fires.

Under this alternative, there would be no salvage logging in the fire stands. Leaving all burned trees in place provides the greatest opportunity to maintain burned habitat for woodpeckers in the long term. Burned trees fall down over time and consequently produce abundant downed woody materials.

Alternative 2. Salvage all burned stands with qdbh greater than 15 cm.

Alternative 2 would remove the special burned wildlife habitat, but increase total timber harvest levels for forest companies.

Alternative 3. Salvage some burned stands with qdbh greater than 15 cm.

Based on the range of burned areas and the fact that there is a greater frequency of small fires, I define salvage areas of 2,000 ha, 5,000 ha, 10,000 ha and 20,000 ha as threshold values for salvage action. If the burned area each year is less than the defined threshold value, no salvage operation occurs, and burned trees are left in the forest stands; or, if the burned area each year is larger than the defined threshold, salvage logging occurs only in the area larger than the threshold (Figure 2.2). Different thresholds are designed to reflect the potential influences of remaining burned area on maintaining burned habitat for woodpeckers. In the rest of this paper, I designated thresholds of 2,000 ha, 5,000 ha, 10,000 ha, and 20,000 ha as "th2000", "th5000", "th10000", and "th20000"; "salnone" indicates no salvage logging in the burned stands, and "salall" indicates salvaging all burned stands.

While salvaging all burned trees may negate their important ecological functions, leaving all fire-killed trees results in a decrease in timber recovery and financial revenues. These alternatives were developed as a potential strategy to address the conflicting demands of the forest industry and forest conservation in terms of timber recovery, financial revenue, and burned habitat maintenance.

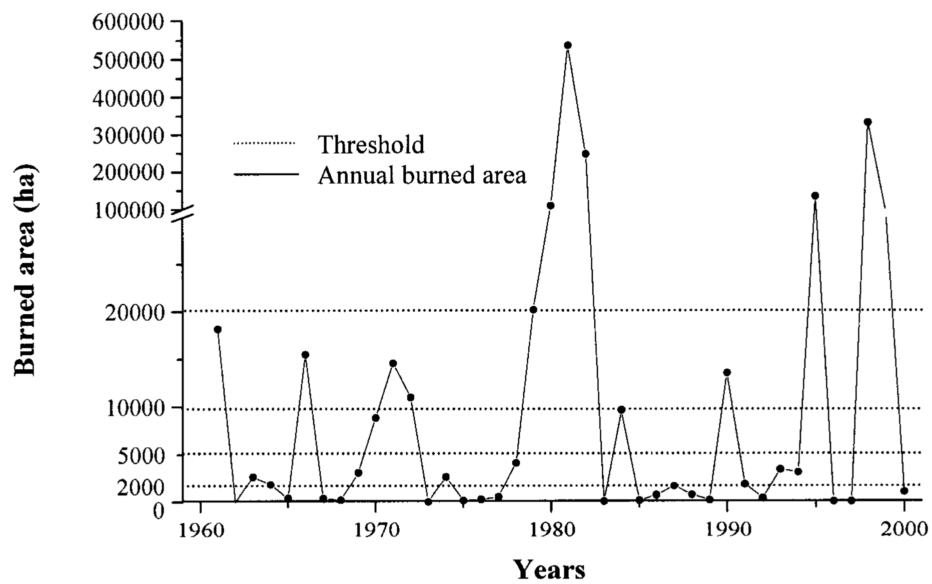


Figure 2.2. Illustration of the thresholds of 2,000, 5,000, 10,000, and 20,000 ha. The salvaged area each year is the area above the defined threshold value.

In keeping with the salvage logging alternatives, three treatment scenarios were assumed for both salvaged and unsalvaged stands.

<u>Scenario 1.</u> Neither salvaged nor unsalvaged stands were reforested, and both of them were deducted from the productive landbase in the determination of AACs.

<u>Scenario 2.</u> Salvaged stands were reforested, and unsalvaged burned stands were deducted from the productive landbase.

<u>Scenario 3.</u> Salvaged stands were reforested, and unsalvaged burned stands were assumed to regenerate naturally, i.e. that deciduous stands would regenerate immediately after fire disturbances, and coniferous stands would be considered productive in 20 years after fires.

Scenarios 1 and 2 were interpreted to represent the current policy. That burned stands were deducted from the productive landbase is an extreme assumption, but it is consistent with current salvage and AAC determination policies (SRD 2002). Scenario 3 represented the biological concerns that burned areas will regenerate naturally and become productive stands after a certain number of years. Twenty years is an estimated number to compromise the situations that some stands regenerate quickly, while some may regenerate proportionally or may not regenerate at all. This assumption is more realistic in terms of current forest production and management planning.

The model was conducted with following assumptions:

i. All burned stands with qdbh greater than 15 cm were salvageable, and became subject to salvaging practices and habitat area calculations. Therefore, wherever salvaged volumes or burned habitats are mentioned in what follows, they are related to the burned stands of qdbh greater than 15 cm. Burned stands with qdbh less than 15 cm were not considered to be salvaged or to be habitat in this study.
ii. Fire severity was the same for all fires.
iii. There was no loss in merchantable volume for the burned timber
iv. Natural regeneration in unsalvaged stands assumed that deciduous species in the unsalvaged stands regenerated naturally immediately after fires, and conifers regenerated back to the productive landbase after 20 years.

This model was developed by combining a Monte Carlo simulation of a forest fire with an optimization-based forest harvest model. Over a 200-year planning horizon, a linear program was used to determine AACs under current harvest frameworks and constraints put into practice in the Alberta-Pacific FMA area. The detailed simulation steps were as follows:

Step 1. Set the harvest model and generate an MPS format file.

Based on harvest practices in the Alberta-Pacific FMA area, the harvest model was constructed using the Woodstock Forest Modeling System (Remsoft Inc. 2000). The program generated a linear programming matrix – an MPS matrix – as an input to solve the optimal harvest scheduling using the linear programming solver MOSEK (EKA 2001) in step 2.

Step 2. For each of the 1000 draws:

(a) For each successive year of the 200-year horizon of the simulation:

i. Determine AAC, which represents the timber supply.

ii. Apply the fire regime model, simulating burns in a proportion of forest areas.

iii. Apply different salvage logging alternatives.

iv. Apply the treatment scenarios in terms of reforestation, no reforestation or natural regeneration in burned stands.

v. Calculate timber harvest level including green and burned wood, NPVs and habitat areas.

vi. Update the forest inventory to reflect growth, harvest, fire, and regeneration, and update the MPS matrix.

Harvest scheduling model calculation, incorporating the stochastic fire regime, was run through in MATLAB® (Math Works 2001) and solved with the MOSEK (EKA 2001) optimization toolbox for MATLAB.

2.4 Results and Discussion

Fire is a random process. The fire model, which randomly generated burn rates from the fire lognormal distribution, reflected the Monte Carlo simulation technique. Outcomes of projected timber supply, NPVs, and burned habitat areas were presented as probability distributions. Selecting given probability distributions allows decision makers to base decisions on their attitudes towards risks (Armstrong 2000). Adopting strategies for sustainable forest management requires securing information on economic and ecological outcomes that arise from policy decisions and management actions at the level of the landscape.

AAC

Figure 2.3 presents the projected timber harvest level, including green and burned timber, under Scenario 1. High fire years create more juvenile forests across the landscape, shifting forest age distributions, and consequently affecting forest productivity. Salvaged volumes contribute greatly to the distribution of quantile 0.975 of total harvested volumes, but slightly influence the distributions of quantiles 0.025, 0.25, 0.5, and 0.75. Therefore, the quantile lines, excepting the quantile 0.975, present as well the green timber harvest levels, i.e. the AACs. Obviously salvaged volumes differ with salvage thresholds. All panels show similar distributions of AACs under different salvage logging alternatives in Scenario 1, since burned stands are deducted from the productive landbase each year.

Figure 2.4 illustrates the distributions of projected timber harvest levels, including green and burned timber, under Scenario 2. Because salvaged volumes contribute to the distribution of quantile 0.975 of total timber harvests, causing the volume to reach much higher than other quantiles. In the first panel, the information relates to the alternative of removing all fire-damaged stands and of reforesting afterward. Reforestation practices keep the productive landbase stable, because planted stands are considered as returns to the productive landbase and so become included in the AAC determination. However, fire alters the forest age structure and results in minor reductions (3-12%) in the median of AACs at the end of the simulation year, as compared to current AACs. The last panel represents the scenario where burned stands are left in place for ecological reasons, and these areas get deducted from the productive landbase. There is a 50% probability that the AACs at the end of the simulation year is between 50-70% of the current AAC, and a 2.5% probability that the AAC is below 50% of current AACs. There is a difference of 40% in median AACs at the end of 200 years, between the conditions of salvaging all burned stands and of salvaging noting at all. Other panels show AAC projections in different salvage thresholds – 2,000 ha, 5,000 ha, 10,000 ha, 20,000 ha – the distributions of which are all ranged between levels of salvaging nothing and of salvaging all burned stands.

Figure 2.5 shows distributions of projected timber harvest levels, including green and burned timber, under Scenario 3. The salvaged volumes present a pattern of decline corresponding with the increases in salvage thresholds. They are similar in this regard to other treatment scenarios, as

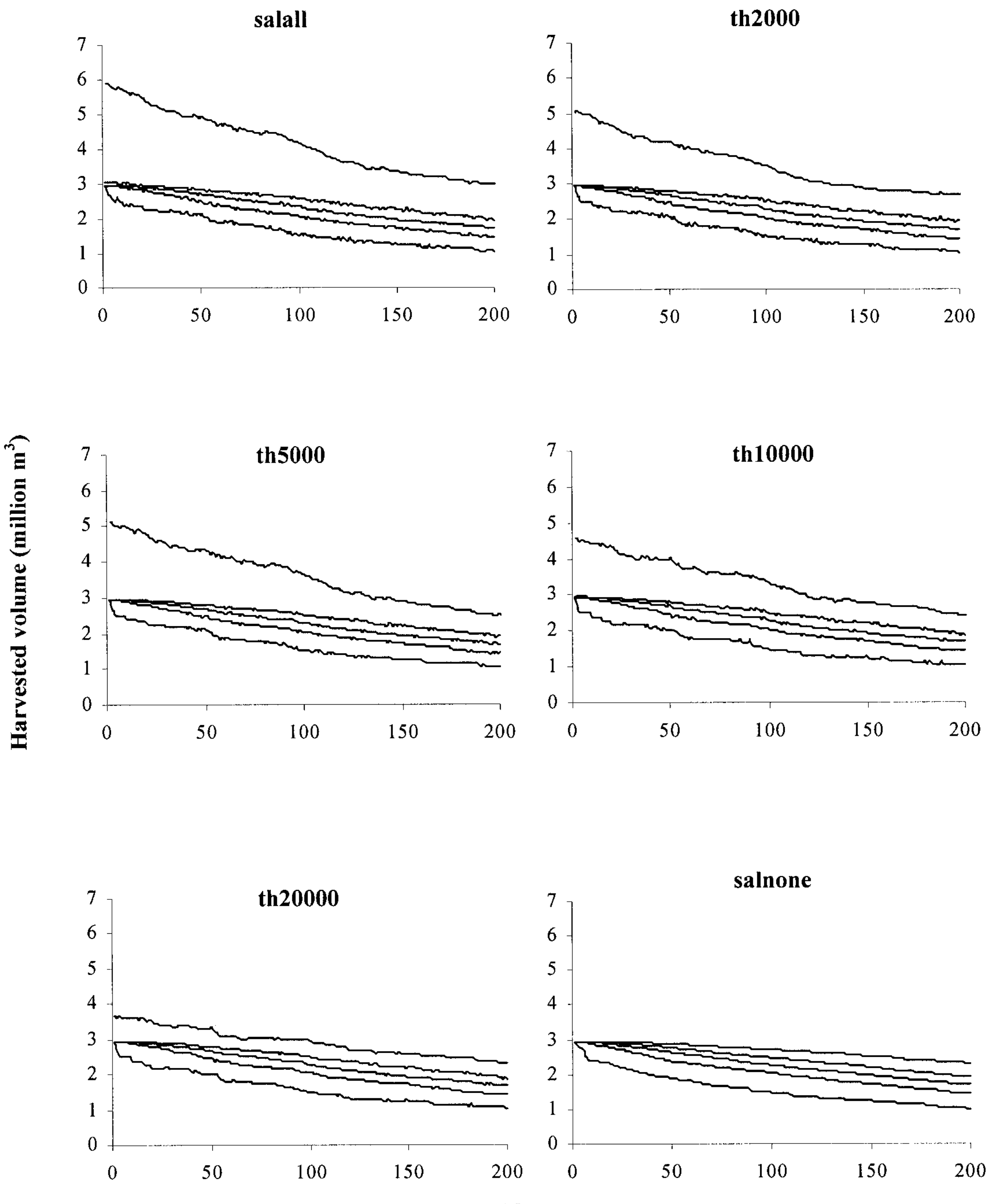


Figure 2.3. Projected distribution of volume harvested (green and burned timber) under different salvage logging alternatives subject to Scenario 1. The panel titles indicate the alternatives, i.e. "salall" indicates salvaging all burned stands; "th2000", "th5000", "th10000", and "th20000" indicate salvaging under each threshold; and "salnone" indicates salvaging nothing. The lines, from bottom to top, represent the 0.025, 0.25, 0.50, 0.75, and 0.975 quantiles.

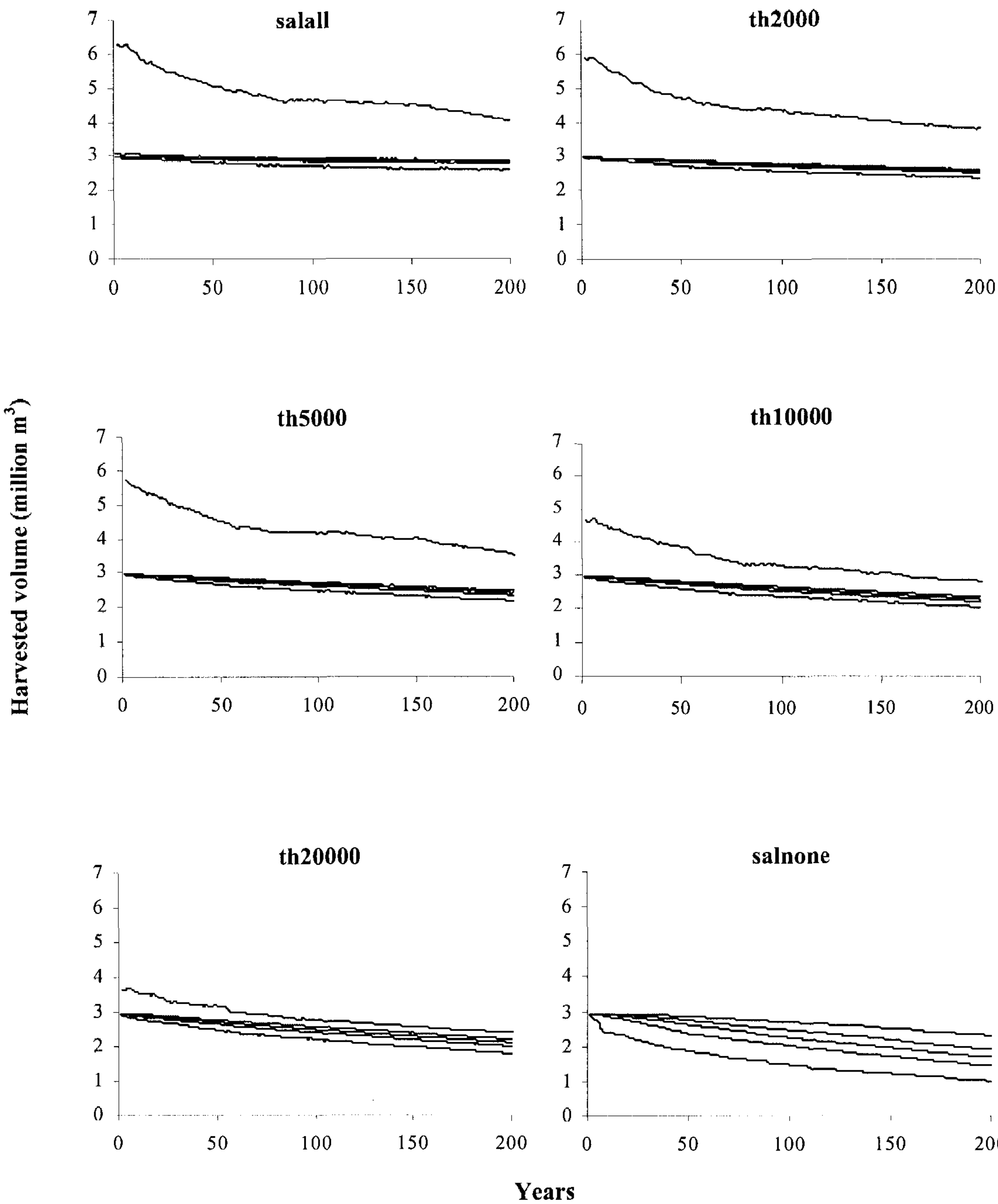


Figure 2.4. Projected distribution of volume harvested (green and burned timber) under different salvage logging alternatives subject to Scenario 2. The panel titles indicate the alternatives, i.e. "salall" indicates salvaging all burned stands; "th2000", "th5000", "th10000", and "th20000" indicate salvaging under each threshold; and "salnone" indicates salvaging nothing. The lines, from bottom to top, represent the 0.025, 0.25, 0.50, 0.75, and 0.975 quantiles.

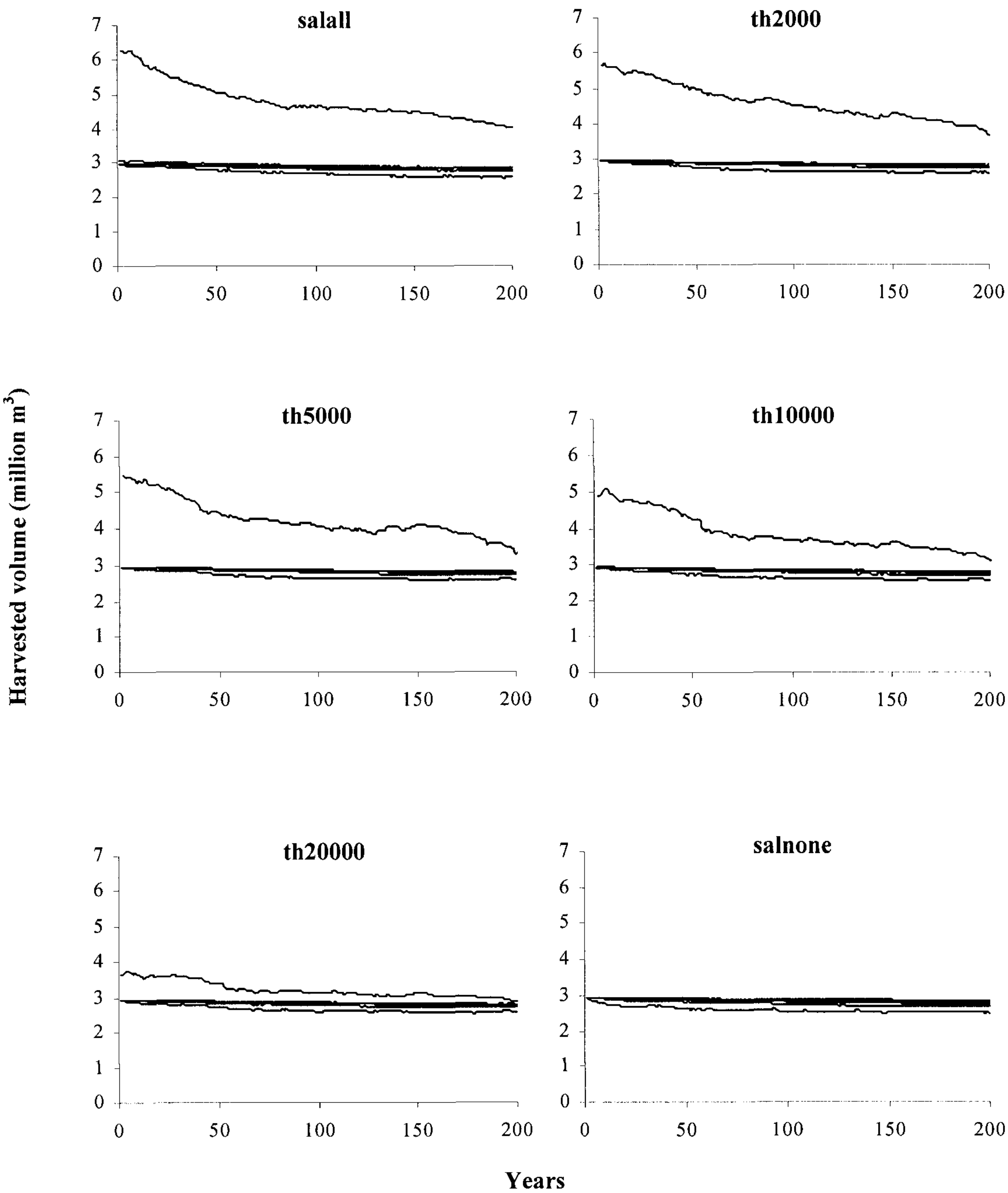


Figure 2.5. Projected distribution of volume harvested (green and burned timber) under different salvage logging alternatives subject to Scenario 3. The panel titles indicate the alternatives, i.e. "salall" indicates salvaging all burned stands; "th2000", "th5000", "th10000", and "th20000" indicate salvaging under each threshold; and "salnone" indicates salvaging nothing. The lines, from bottom to top, represent the 0.025, 0.25, 0.50, 0.75, and 0.975 quantiles.

evidenced in Figures 2.3 and 2.4. However, there are very small variations in median AACs – only a 5% difference at the end of 200 years – between the divergent alternative of salvaging all or salvaging none of the burned stands.

NPV

Figure 2.6 shows the NPV boxplot distributions, as well as NPV_r from green timber and NPV_b from burned timber, under different scenarios. That fires are highly variable results in the large deviations in the probability distributions of NPV_b from salvaged timber. Among the three scenarios, Scenario 3 provides the highest values of NPV for all salvage logging alternatives, and Scenario 1 provides the lowest. For each scenario, removing all fire-killed stands provides the highest value return, whereas leaving them has the lowest, with differences in median NPV of approximately 8.2% in Scenario 1, 9.7% in Scenario 2, and 5.4% in Scenario 3.

The graph also illustrates that the NPV_b in Scenario 1 is higher than that in other two scenarios, ranging from 22% higher in median when all burned stands get salvaged, to 0 with no salvage. In Alberta, reforestation investment offers negative financial returns, due to slow yield and growth rates, long rotation periods, and low product prices. As shown in Tables 2.7 and 2.8, the conversion return is 16.06 C\$/m^3 for coniferous and 11.83 C\$ /m^3 for deciduous species, and the cost of reforestation in harvested stands is 920 C\$/ha and 1012 C\$/ha in salvaged stands. The discount rate is assumed to be 0.05. Table 2.9 shows soil expectation values (SEVs) in regular harvested and salvaged stands over an unbounded time period. The SEVs in coniferous stands are negative, which suggests that forest companies are likely to salvage burned stands and leave them to regenerate naturally. However, through the allowable cut effect (ACE) analysis (Davis et al. 2001), intensive silviculture practices would increase current timber harvest levels and realize the even flow constraint at the forest level by spreading anticipated future growth over years, and would generate positive SEVs.

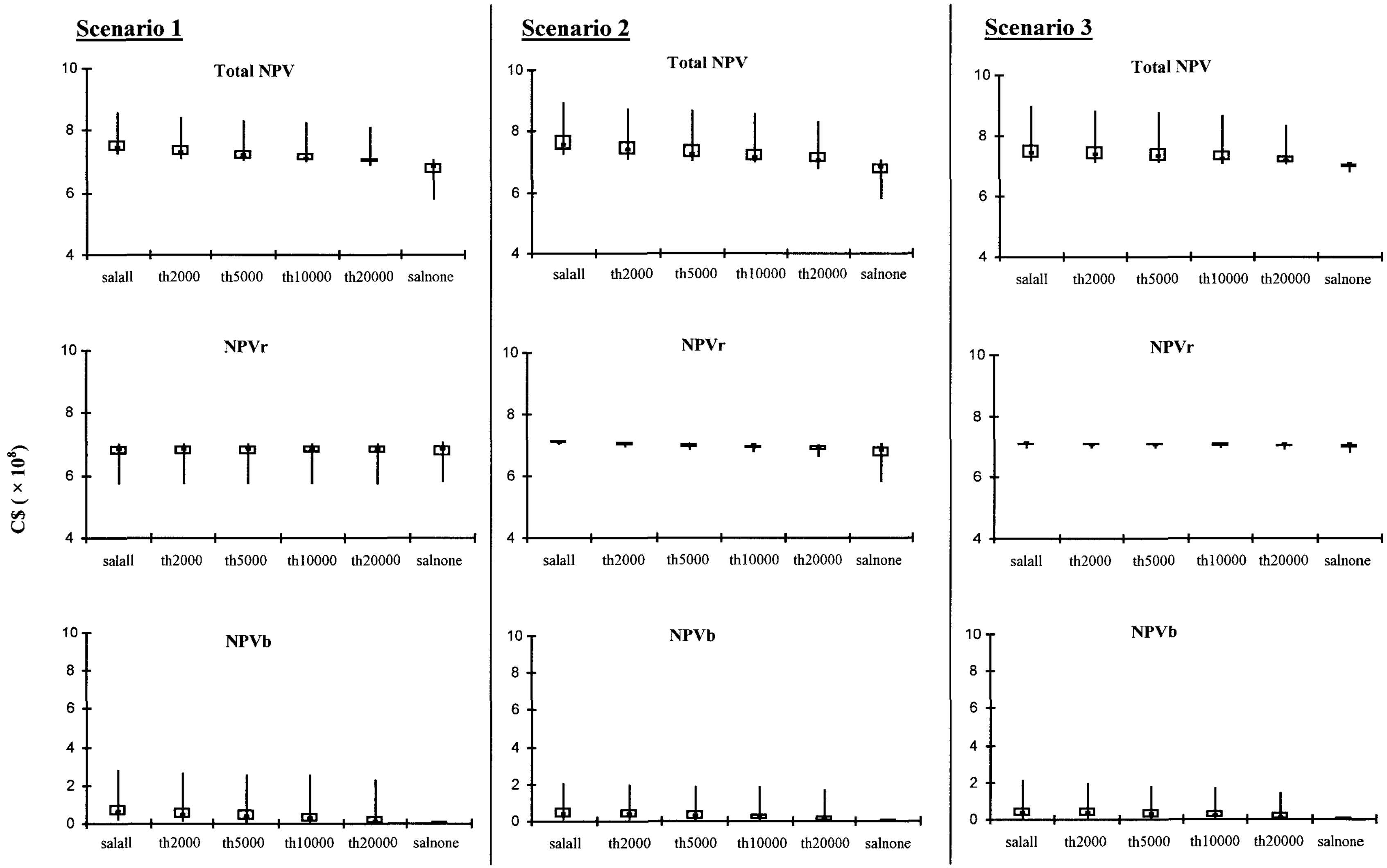


Figure 2.6. Box plots of NPV, NPVr and NPVb under different salvage alternatives and treatments scenarios. Boxes represent 0.25 and 0.75 quantiles with median bars inside. The total range is between 0.025 and 0.975 quantiles. NPVr is the net present value from regular harvested green timber, NPVb is the net present value from salvaged timber, and total NPV is the sum of them.

Table 2.9. Soil expectation value (SEV) based on reforestation investment. TPR represents the timber productivity rating. G, M, and F represent good, median, and fair TPR sites respectively.

Cover type	TPR	Rotation age R (years)	Yield Y (m^3/ha)	Regeneration cost1*(C$/ha)	Regeneration cost2*(C$/ha)	Interest rate (i)	Conversion return Rt (C$/$m^3$)	SEV1** (C$)	SEV2** (C$)
White spruce	G	100	340.93	920	1,012	0.05	16.06	-885.09	-977.80
	M	130	300.75	920	1,012	0.05	16.06	-913.11	-1,005.27
	F	180	250.49	920	1,012	0.05	16.06	-919.52	-1,011.54
Black spruce	G	140	354.47	920	1,012	0.05	16.06	-914.84	-1,006.94
	M	180	236.15	920	1,012	0.05	16.06	-919.56	-1,011.57
	F	180	69.32	920	1,012	0.05	16.06	-919.97	-1,011.98
Pine	G	90	350.12	920	1,012	0.05	16.06	-861.01	-954.17
	M	120	292.95	920	1,012	0.05	16.06	-909.12	-1,001.39
	F	160	210.57	920	1,012	0.05	16.06	-919.00	-1,011.03
Conifer-dominated mixedwood	G	110	278.77	920	1,012	0.05	16. 06	-903.32	-995.75
	M	140	239.48	920	1,012	0.05	16. 06	-916.84	-1,008.94
	F	180	205.95	920	1,012	0.05	16. 06	-919.63	-1,011.65
Deciduous-dominated mixedwood	G	80	140.82	920	1,012	0.05	11.83	-904.64	-998.53
	M	100	124.03	920	1,012	0.05	11. 83	-915.81	-1,008.51
	F	100	68.37	920	1,012	0.05	11. 83	-920.85	-1,013.56
Deciduous	G	70	255.09	0	0	0.05	11. 83	102.55	102.55
	M	90	209.76	0	0	0.05	11. 83	31.12	31.12
	F	100	115.68	0	0	0.05	11. 83	10.49	10.49

* Regeneration cost 1 is the reforestation costs in regular harvested stands.
Regeneration cost 2 is the reforestation costs in salvage harvested stands.

** SEV1: soil expectation value in regular harvested stands. SEV1 = $(Rt \times Y - cost1 \times (1+i)^R) / ((1+i)^R - 1)$

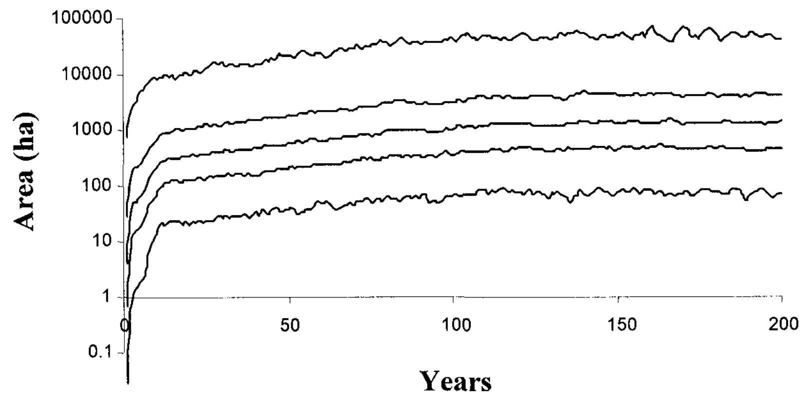
SEV2: soil expectation value in salvage harvested stands. SEV2 = $(Rt \times Y - cost2 \times (1+i)^R) / ((1+i)^R - 1)$

Habitat

With sustainable forest management, biodiversity and habitat considerations are becoming more important in forest management planning. The concern for habitat protection is also an important criterion to be considered during the development of salvage plans. Fire creates burned areas in both passive and productive landbases. Snags in the passive landbase would likely not be salvaged. A number of snags would become available here to maintain certain levels of habitat for wildlife, both immediately after fires and in late succession of burned stands. However, the passive landbase is dominated by tree species of tamarack, jack pine and black spruce, which provide habitat only for certain types of wildlife. Figure 2.7 shows the projected distributions of burned habitat areas for the two woodpecker species in the passive landbase.

In the productive landbase, however, burned stands are subject to salvage harvesting. Burned habitat areas change under different salvage logging scenarios. As Scenario 3 is the situation of more likely occurrence in the burned stands, this section mainly presents its simulation results. Figure 2.8 shows the changes in mean habitat areas for Black-backed Woodpeckers and Three-toed Woodpeckers under different salvage thresholds subject to Scenario 3. All lines show

BBWP habitat in passive landbase



TTWP habitat in passive landbase

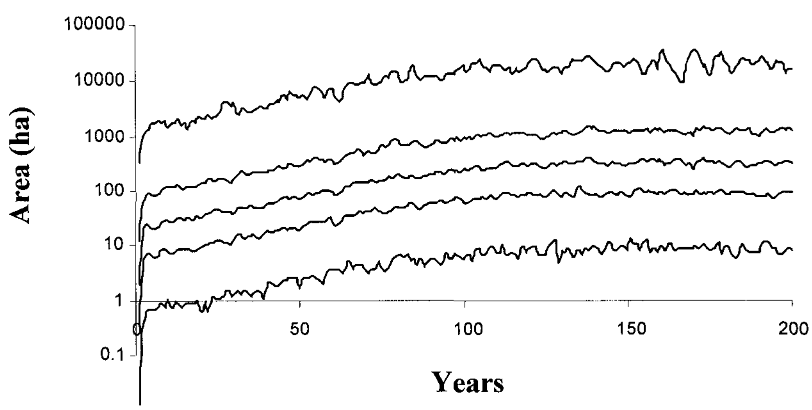
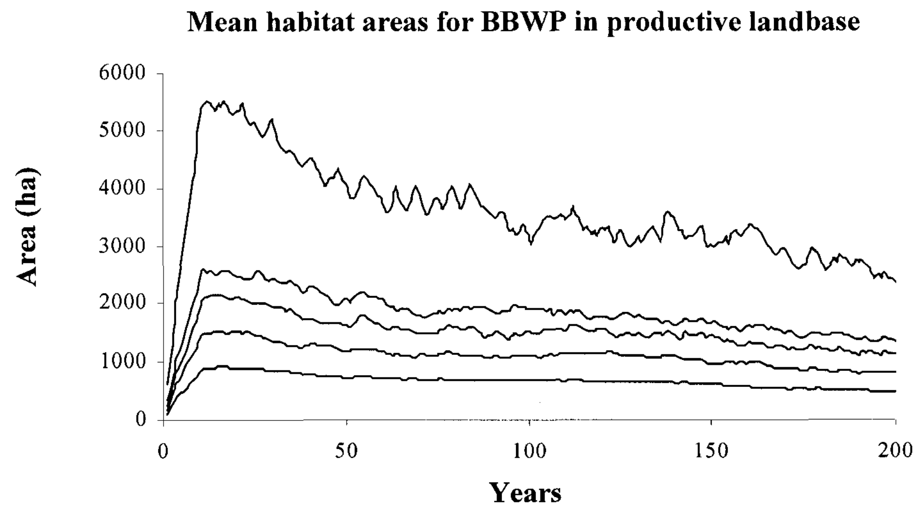


Figure 2.7. Projected habitat quantiles for BBWP and TTWP in the passive landbase. The passive landbase is not subject to salvage logging and harvesting. Quantile lines, from bottom to top, are 0.025, 0.25, 0.5, 0.75, and 0.975. Y-axis is a logarithmic scale.



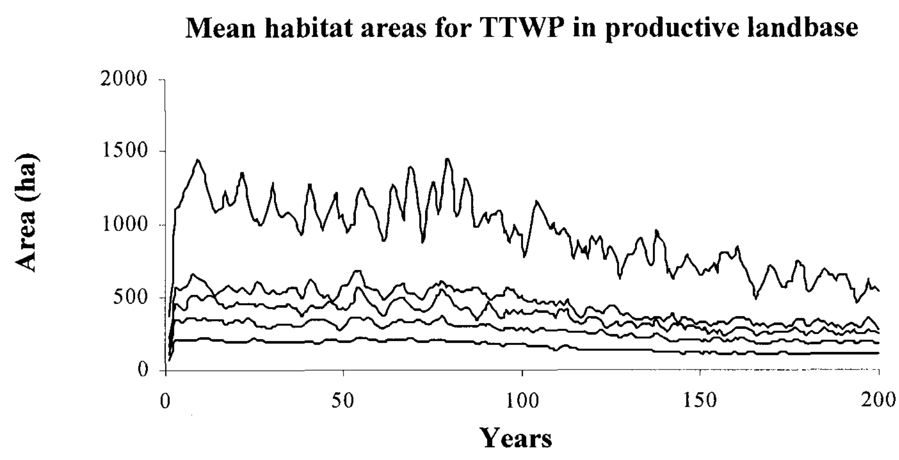


Figure 2.8. Mean habitat areas for BBWP and TTWP in the productive landbase subject to Scenario 3. The lines represent, from bottom to top, the habitat areas with the threshold of 2,000 ha, 5,000 ha, 10,000 ha, and 20,000 ha, as well as with salvaging nothing.

a declining tendency because the forest landscape tends to have more juvenile stands after harvesting and fire disturbances. Therefore, old stands in the productive landbase become gradually fewer.

The trade-offs between timber harvest levels and habitat areas for Black-backed Woodpeckers in year 200 under Scenarios 1 and 2 are presented in Figures 2.9 and 2.10. Figures 2.11 through 2.15 show the tradeoffs in years 25, 50, 100, and 200 under Scenario 3. These graphs reveal that among the salvage alternatives, Scenario 3 generates higher timber harvest levels and larger habitat areas than Scenario 2, and Scenario 2 in turn generates higher ones than Scenario 1. The assumptions of reforestation after salvaging and natural regeneration in the unsalvaged stands contribute to the return of burned stands back to the productive landbase and thus positively affect AAC determination. More productive landbase also contributes to the enhanced production of burned habitat after fires. With the increase in the threshold value, the mean habitat area increases, but timber harvest level decreases. However, after considering the influences of different thresholds on the mean harvested volumes, there are large differences in the harvest levels with the increase in the thresholds in the long term among the scenarios (Figures 2.9, 2.10, and 2.14). For Scenario 3, increasing salvage threshold does not substantially affect the median of harvest levels, but provides larger value of mean habitat areas. Therefore, leaving certain amounts of burned areas for wildlife in a forest causes slight changes of median harvested volumes in the long term. Harvest levels present little risks on downside but substantial risks on the upside of median harvested volumes. Salvaging after fires provides high probabilities for forest companies to obtain more timber volumes. These trade-off graphs would assist forest managers to determine appropriate salvage logging rates, taking both goals – timber extraction value and wildlife habitat preservation – into consideration.

The useful outputs, under Scenario 3, of burned habitat by stage and cover type are given in Figures 2.15 through 2.19. Obviously, there are major changes in mean habitat areas in three stages with the different salvage thresholds. The larger the threshold is, the more habitat areas are provided for wildlife, and the alternative of salvaging nothing provides the most. As the simulation began from year 0, and as it was assumed that there were no burned stands prior to the starting year, all habitat areas derived from the simulation indicate 0 ha in year 0. Numerous studies have found that there were major differences in bird communities in burned forests with regard to cover types and age structures (Hobson and Schieck 1999, Saab and Dudley 1998, Morissette et al. 2002). Newly burned stands are dominated by standing snags, which attract

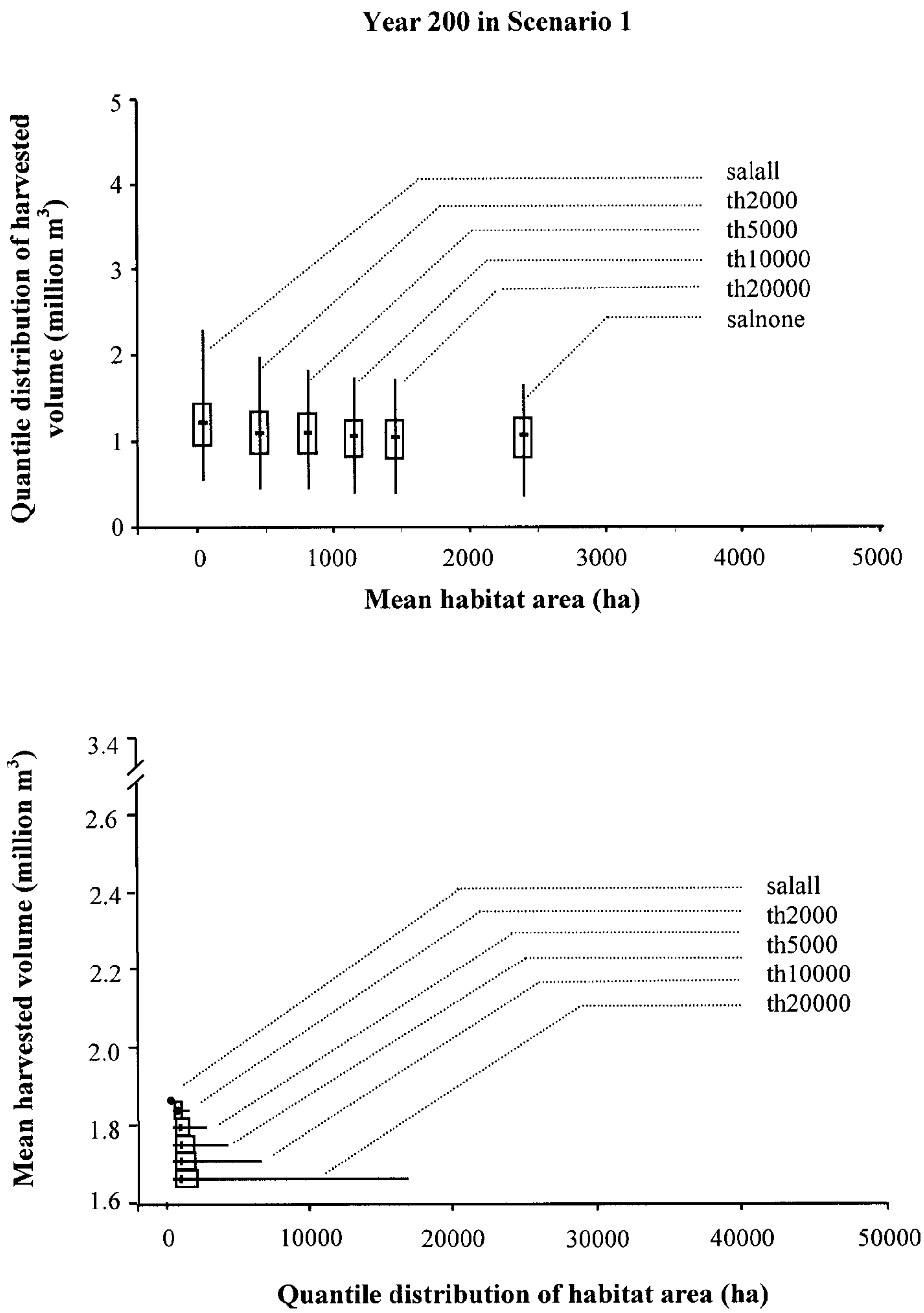
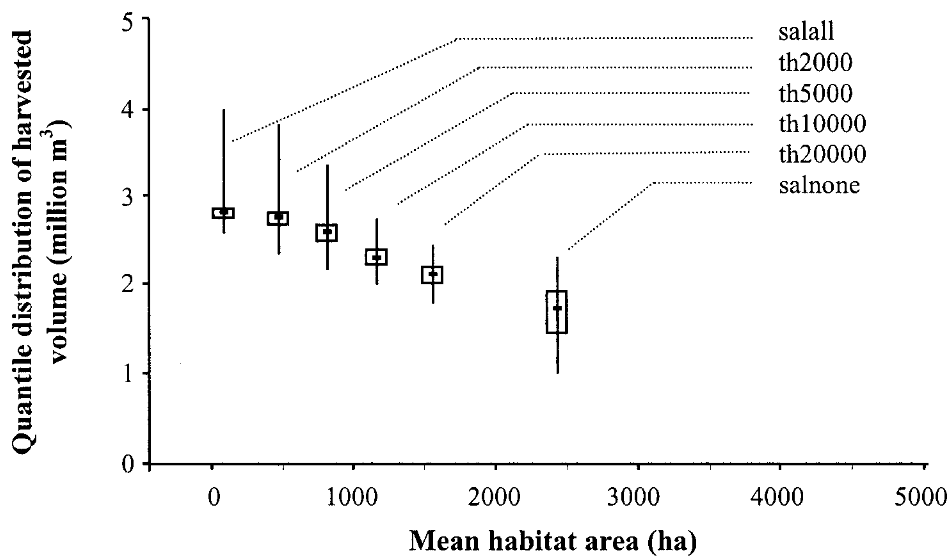


Figure 2.9. Tradeoffs of harvested volumes and habitat areas in year 200 subject to Scenario 1. Volumes are presented by quantile distributions of 0.025, 0.25, 0.5, 0.75, and 0.975 from bottom to top. Habitat areas are presented by quantile distributions of 0.025, 0.25, 0.5, 0.75, and 0.975 from left to right. "salall" indicates salvaging all burned stands; "th2000", "th5000", "th10000", and "th20000" indicate salvaging under each threshold; and "salnone" indicates salvaging nothing.

Year 200 in Scenario 2



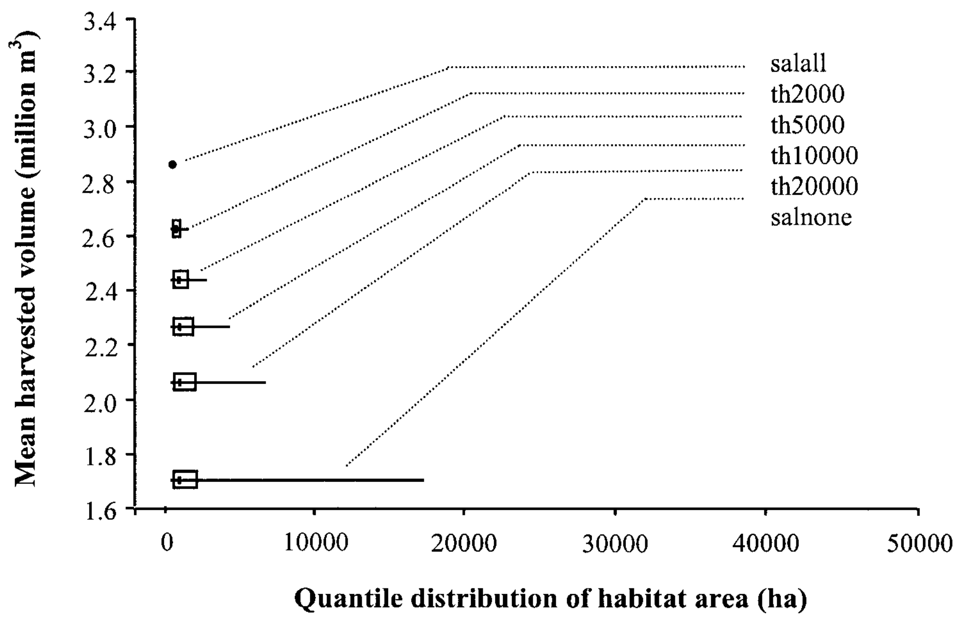


Figure 2.10. Tradeoffs of harvested volumes and habitat areas in year 200 subject to Scenario 2. Volumes are presented by quantile distributions of 0.025, 0.25, 0.5, 0.75, and 0.975 from bottom to top. Habitat areas are presented by quantile distributions of 0.025, 0.25, 0.5, 0.75, and 0.975 from left to right. "salall" indicates salvaging all burned stands; "th2000", "th5000", "th10000", and "th20000" indicate salvaging under each threshold; and "salnone" indicates salvaging nothing.

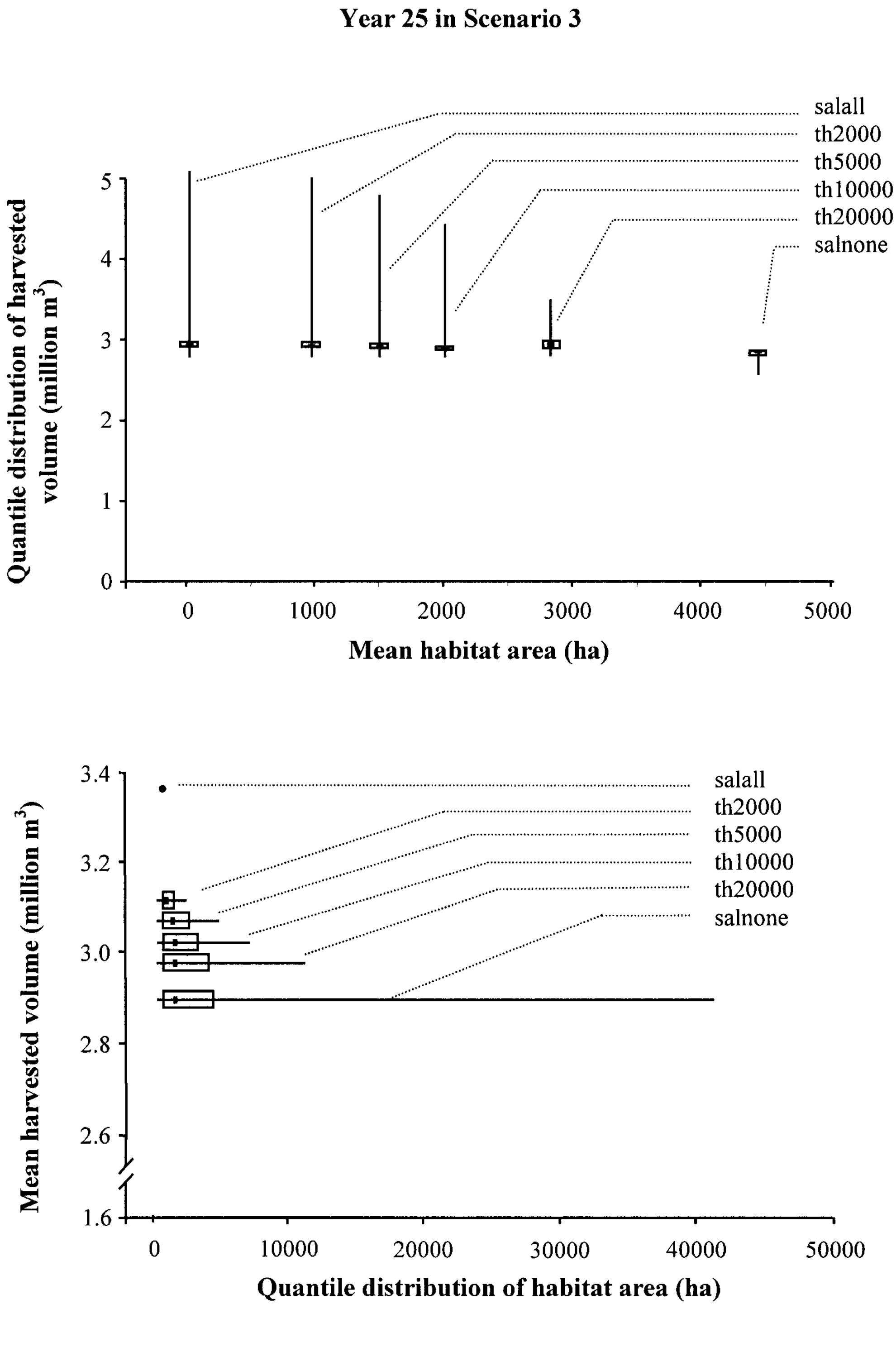


Figure 2.11. Tradeoffs of harvested volumes and habitat areas in year 25 subject to Scenario 3. Volumes are presented by quantile distributions of 0.025, 0.25, 0.5, 0.75, and 0.975 from bottom to top. Habitat areas are presented by quantile distributions of 0.025, 0.25, 0.5, 0.75, and 0.975 from left to right. "salall" indicates salvaging all burned stands; "th2000", "th5000", "th10000", and "th20000" indicate salvaging under each threshold; and "salnone" indicates salvaging nothing.

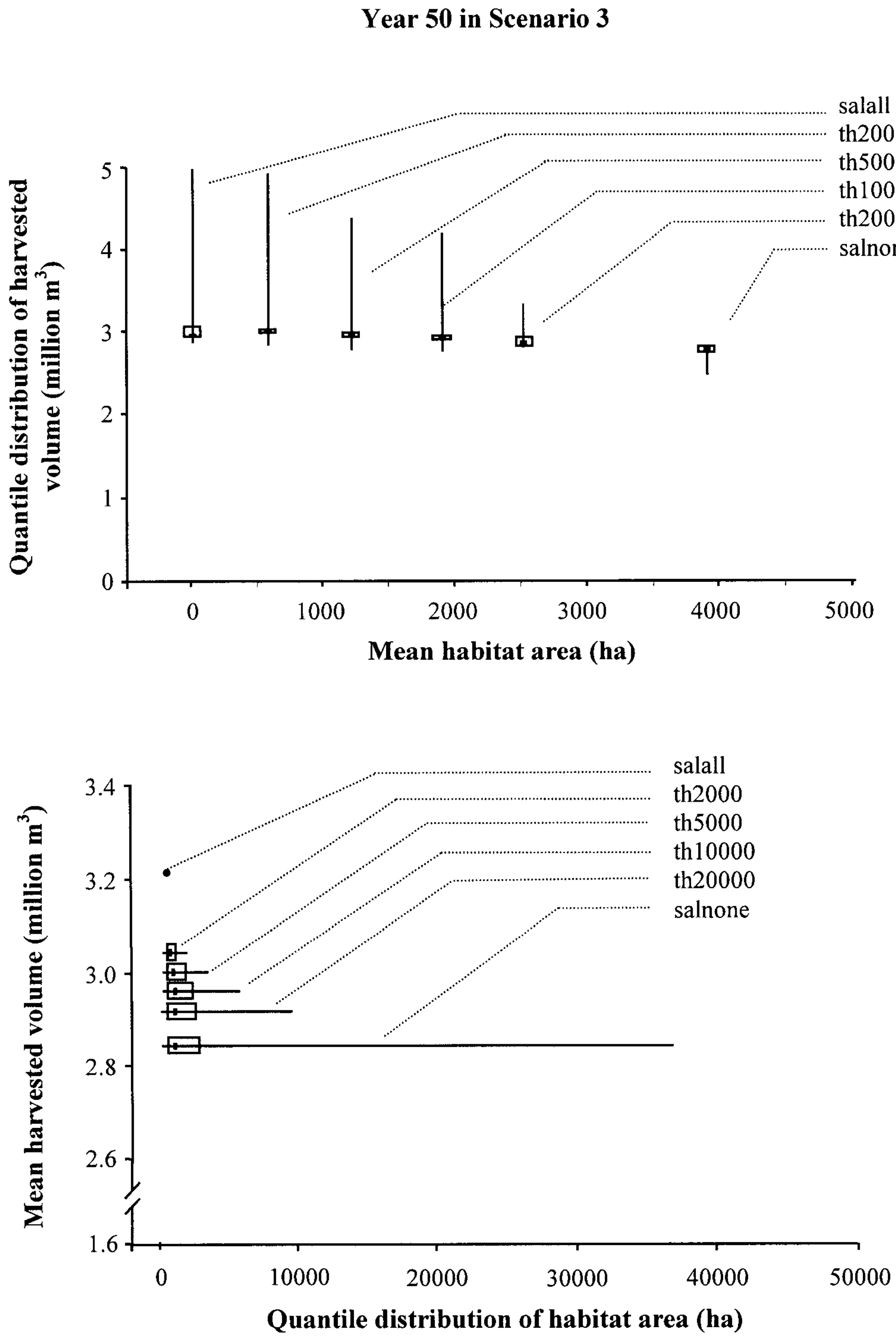


Figure 2.12. Tradeoffs of harvested volumes and habitat areas in year 50 subject to Scenario 3. Volumes are presented by quantile distributions of 0.025, 0.25, 0.5, 0.75, and 0.975 from bottom to top. Habitat areas are presented by quantile distributions of 0.025, 0.25, 0.5, 0.75, and 0.975 from left to right. "salall" indicates salvaging all burned stands; "th2000", "th5000", "th10000", and "th20000" indicate salvaging under each threshold; and "salnone" indicates salvaging nothing.

Year 100 in Scenario 3

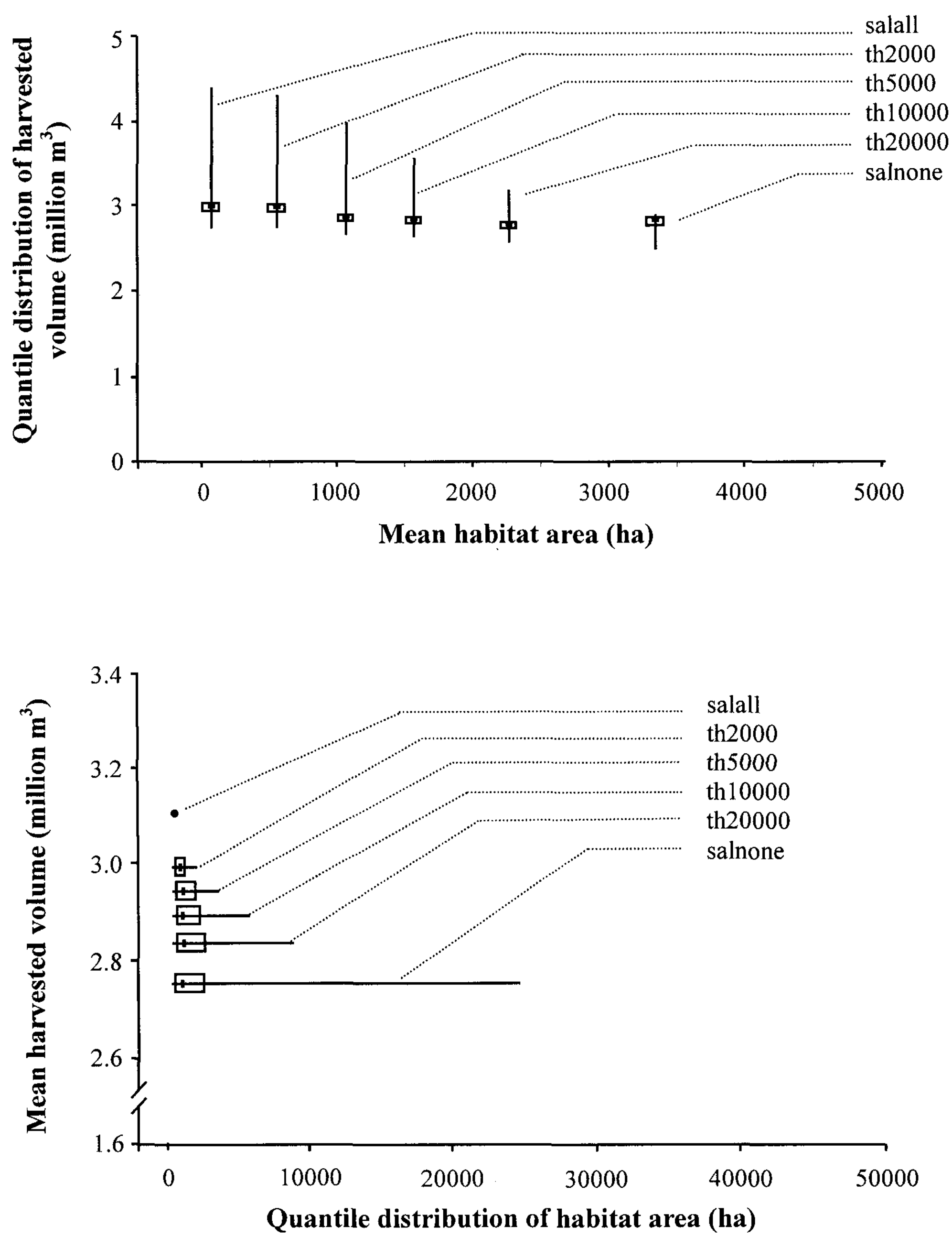
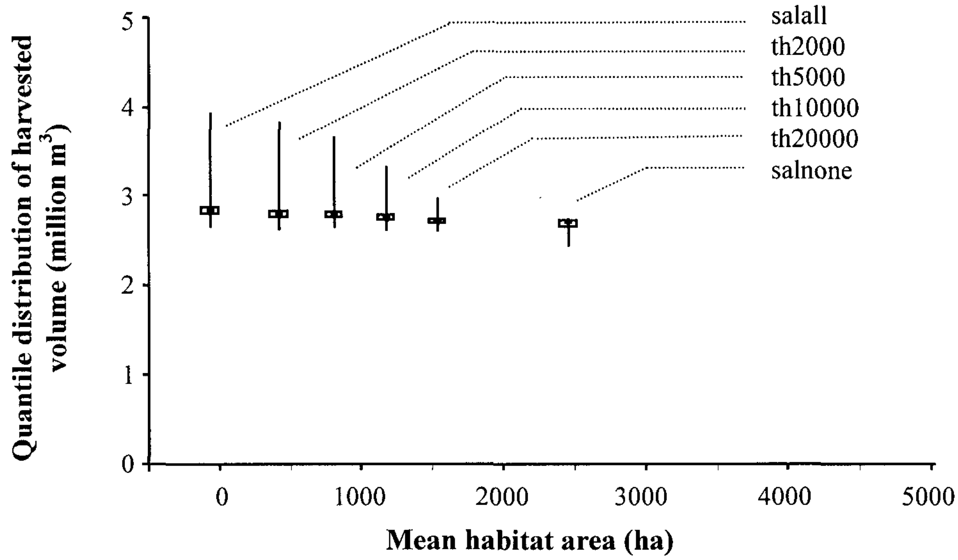


Figure 2.13. Tradeoffs of harvested volumes and habitat areas in year 100 subject to Scenario 3. Volumes are presented by quantile distributions of 0.025, 0.25, 0.5, 0.75, and 0.975 from bottom to top. Habitat areas are presented by quantile distributions of 0.025, 0.25, 0.5, 0.75, and 0.975 from left to right. "salall" indicates salvaging all burned stands; "th2000", "th5000", "th10000", and "th20000" indicate salvaging under each threshold; and "salnone" indicates salvaging nothing.

Year 200 in Scenario 3



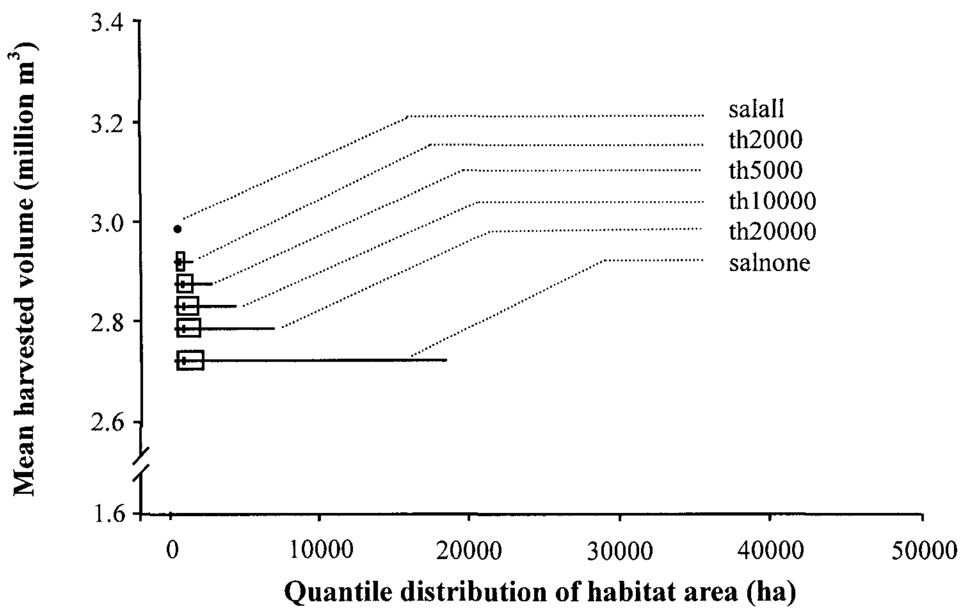


Figure 2.14. Tradeoffs of harvested volumes and habitat areas in year 200 subject to Scenario 3. Volumes are presented by quantile distributions of 0.025, 0.25, 0.5, 0.75, and 0.975 from bottom to top. Habitat areas are presented by quantile distributions of 0.025, 0.25, 0.5, 0.75, and 0.975 from left to right. "salall" indicates salvaging all burned stands; "th2000", "th5000", "th10000", and "th20000" indicate salvaging under each threshold; and "salnone" indicates salvaging nothing.

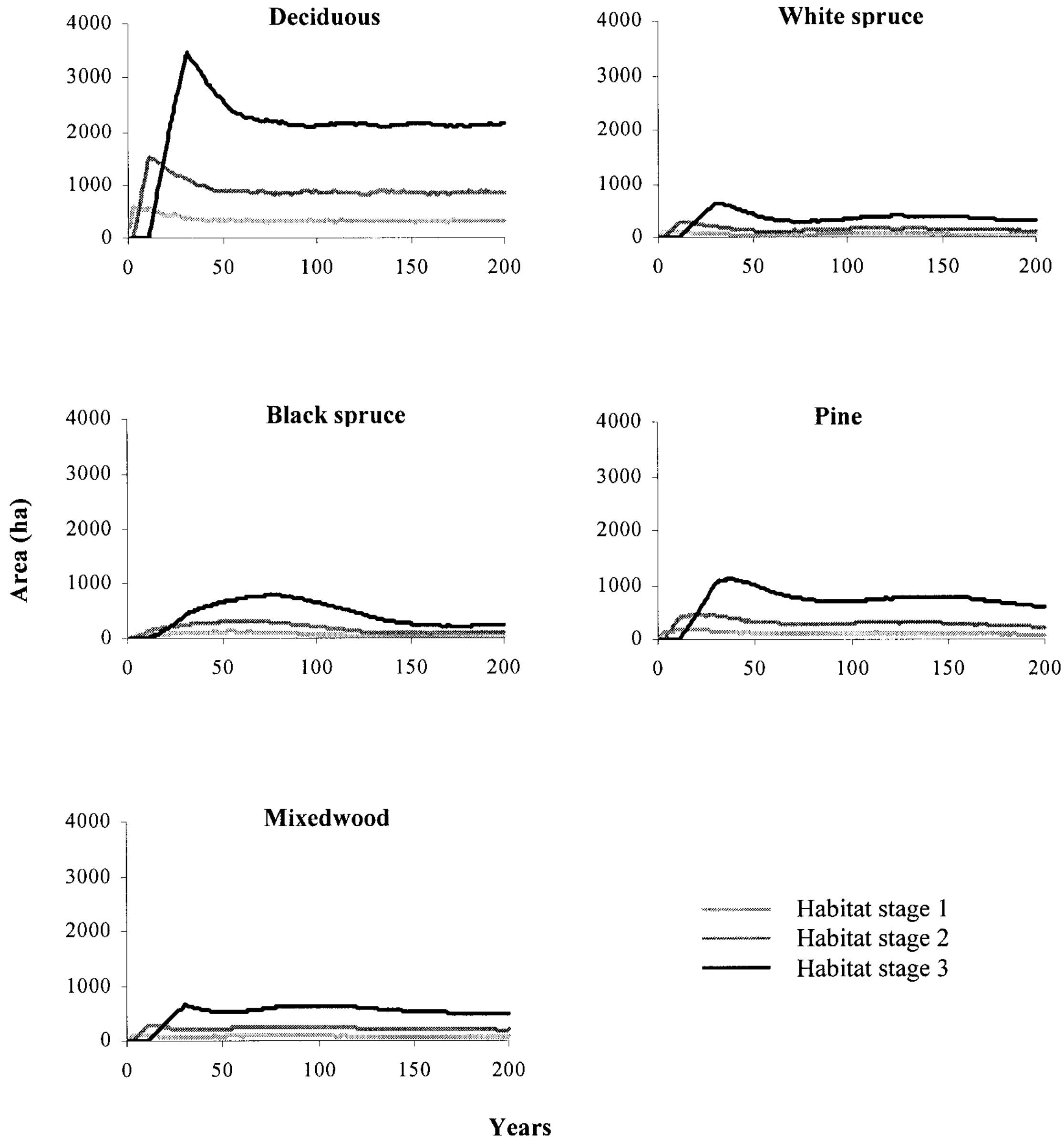


Figure 2.15. Mean habitat areas in different stages by cover types under the salvage threshold of 2,000 ha, subject to Scenario 3.

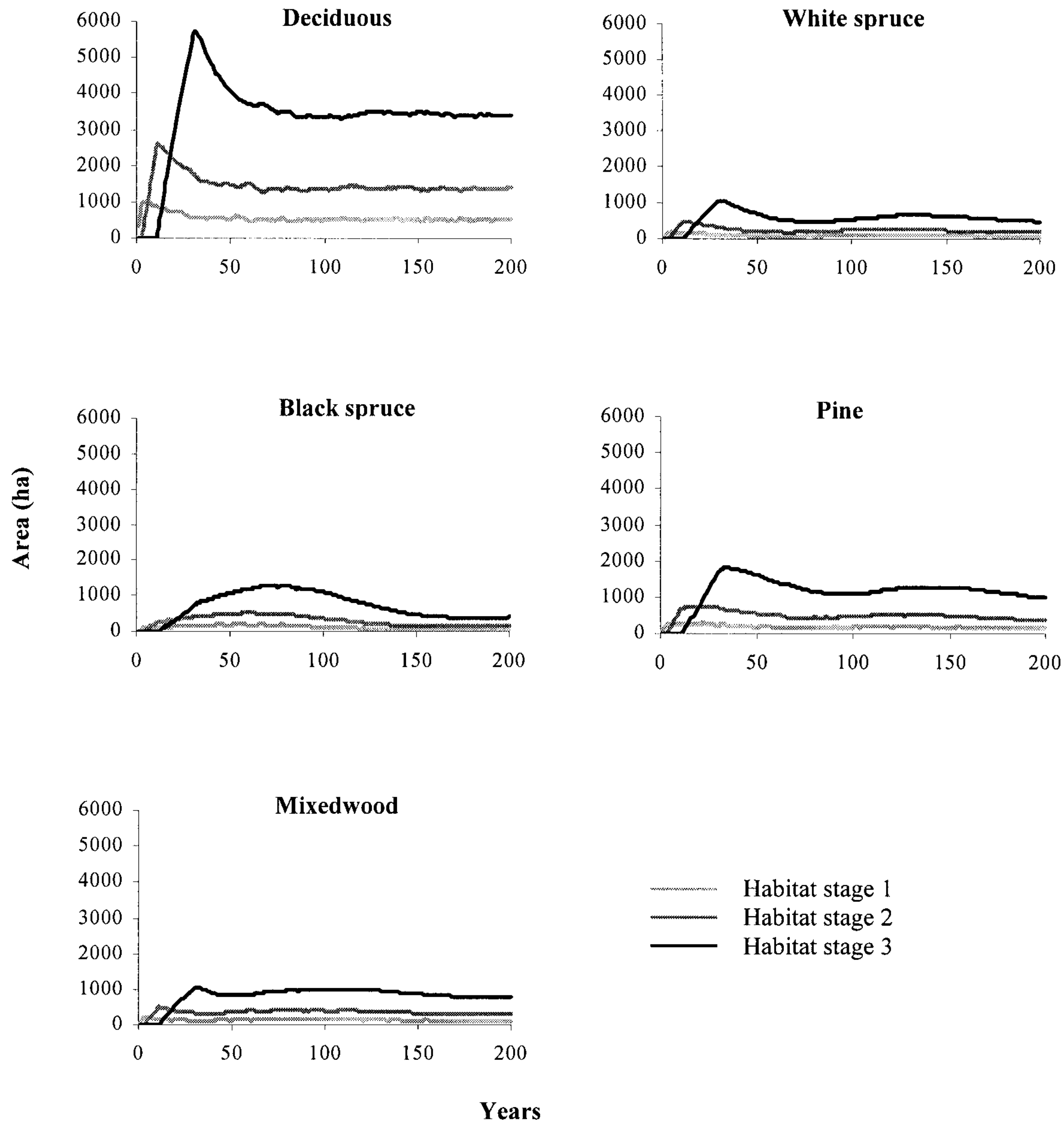


Figure 2.16. Mean habitat areas in different stages by cover types under the salvage threshold of 5,000 ha, subject to Scenario 3.

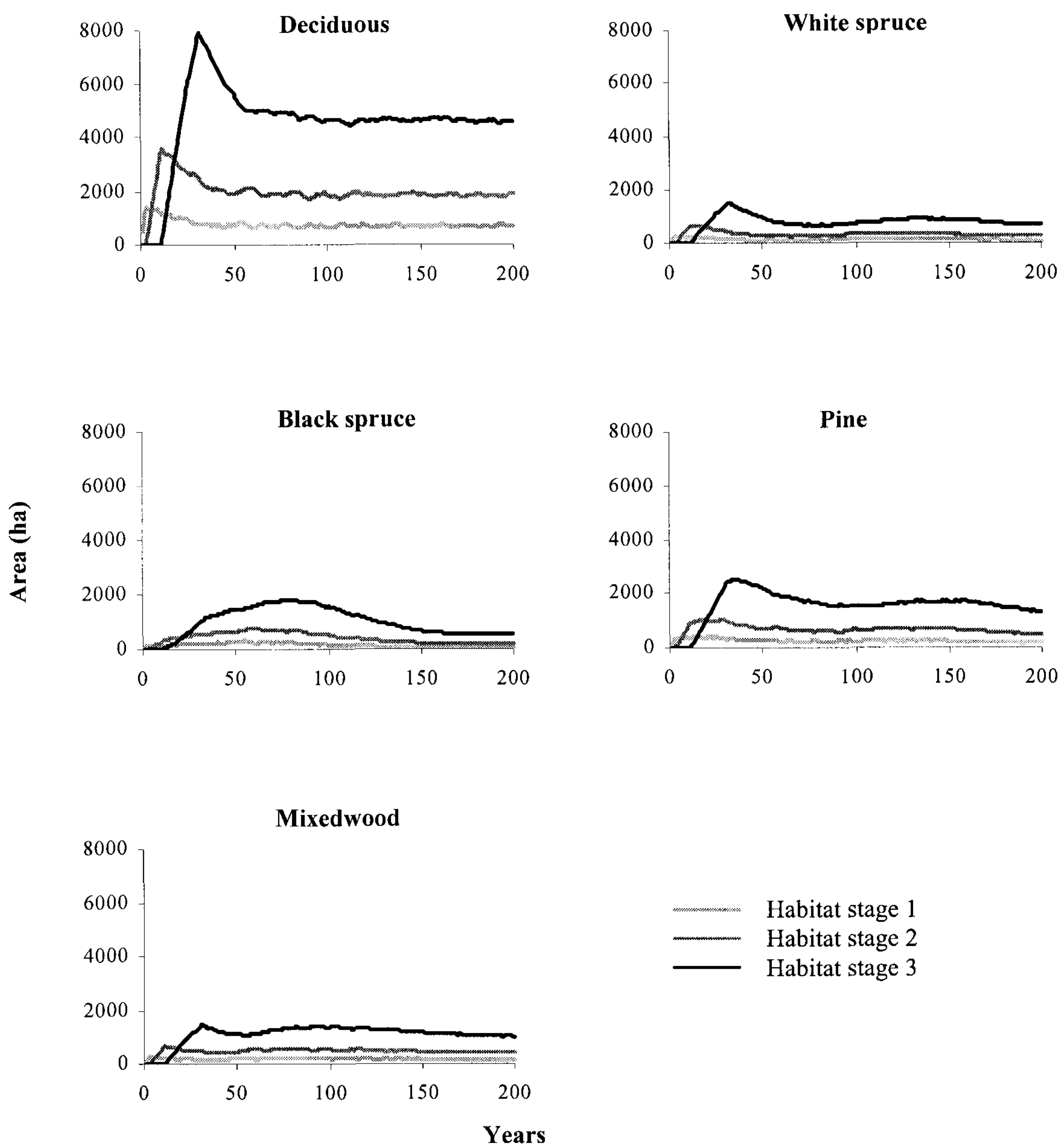


Figure 2.17. Mean habitat areas in different stages by cover types under the salvage threshold of 10,000 ha, subject to Scenario 3.

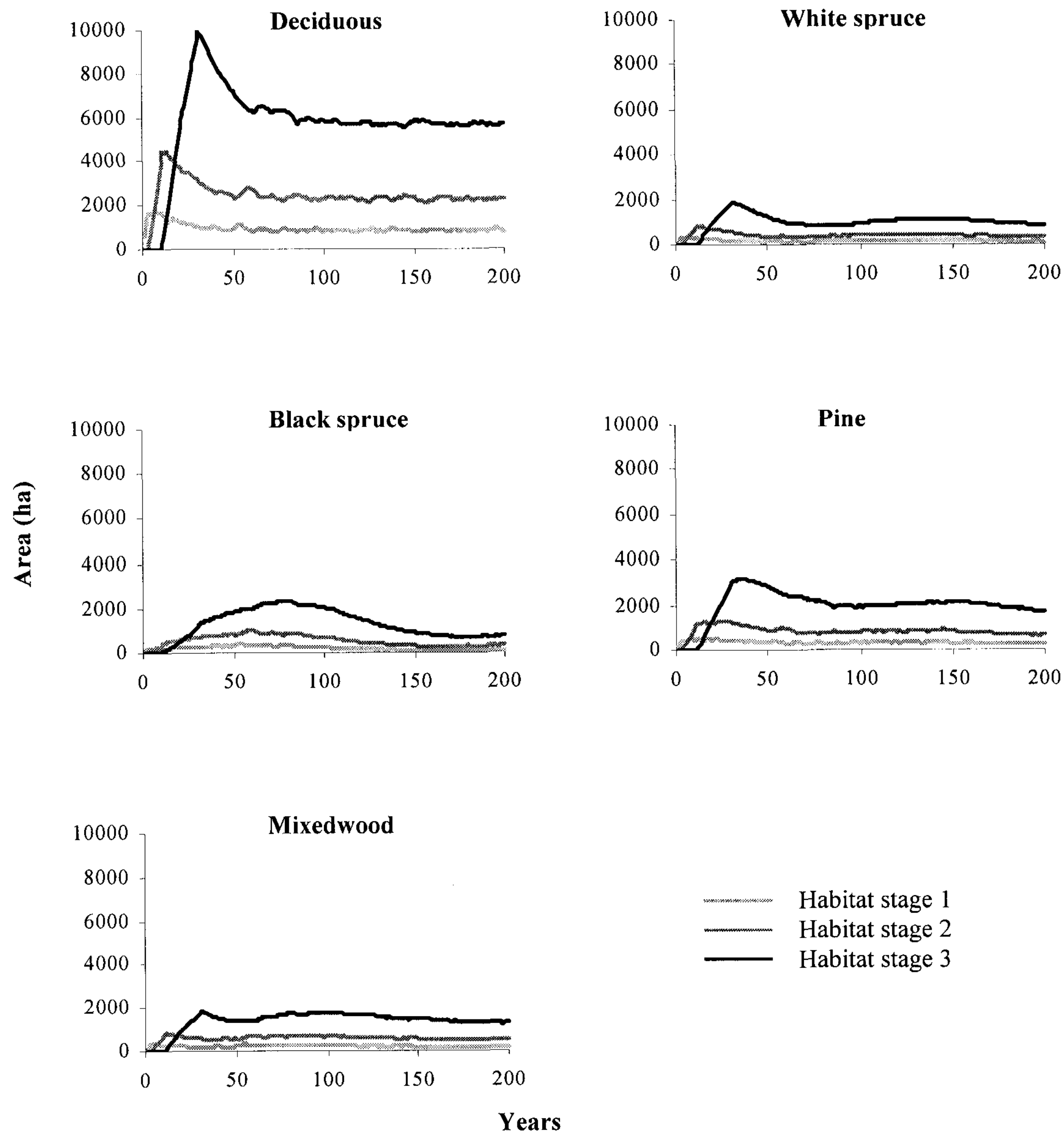


Figure 2.18. Mean habitat areas in different stages by cover types under the salvage threshold of 20,000 ha, subject to Scenario 3.

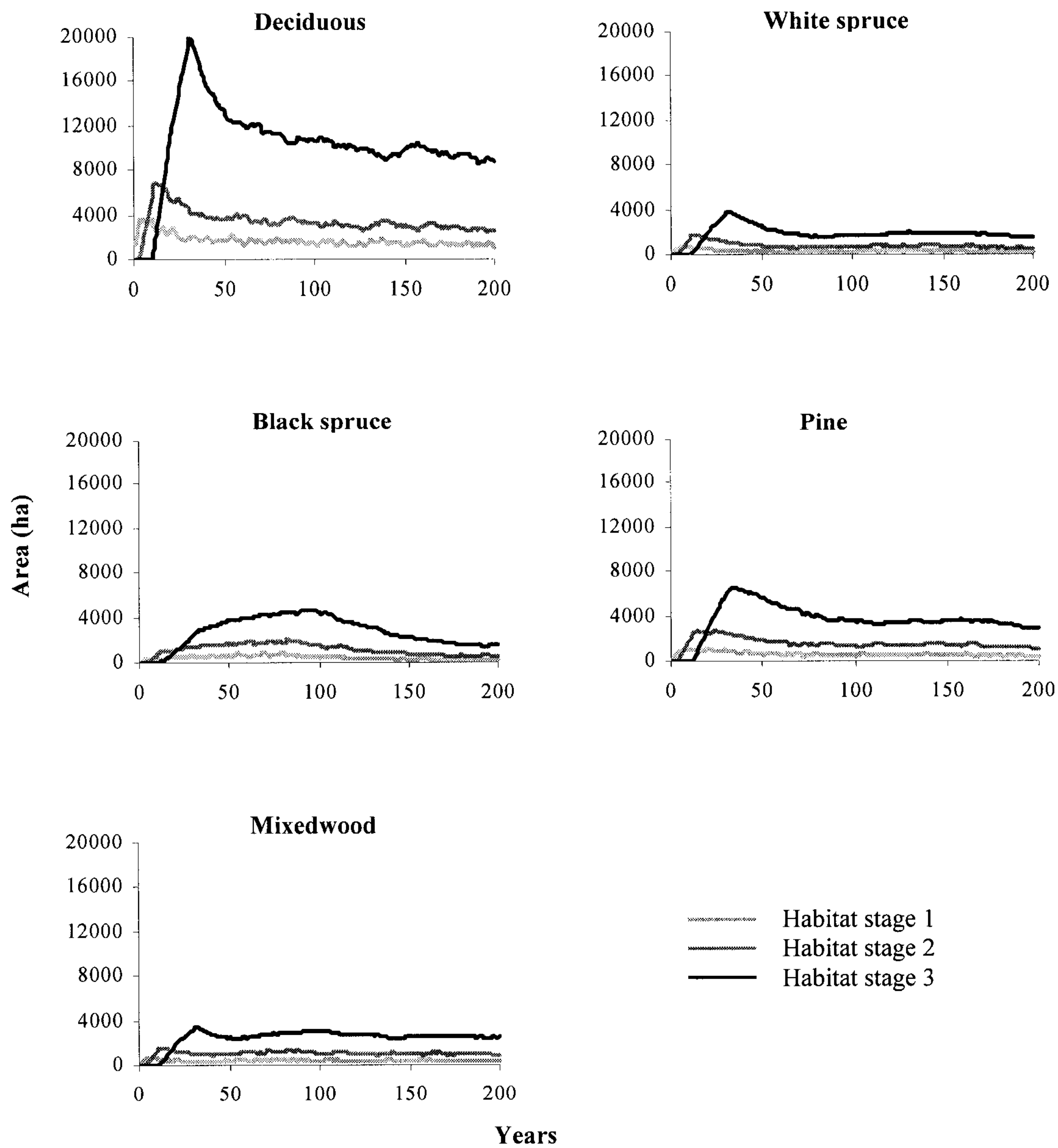


Figure 2.19. Mean habitat areas in different stages by cover types under salvaging nothing, subject to Scenario 3.

insects and woodpeckers, while aging burned stands are dominated by downed woody materials and new vegetation, which are used by some vertebrates. For example, red-backed voles (*Clethrionomys gapperi)*, deer mice (*Peromyscus maniculatus*), and meadow voles (*Microtus pennsylvanicus*) are abundant in stands that are newly burned, as well as in those that are standing several years postfire (Lee 1999, Song 2002). Magnolia Warblers (*Dendroica magnolia*) and Connecticut Warblers (*Oporornis agilis*), by contrast, have high densities in 28-year-old postfire stands (Hobson and Schieck 1999).

2.5 Conclusions

Habitat for Three-toed Woodpeckers or Black-backed Woodpeckers may become reduced by current forestry practices, which include fire suppression, short harvest rotations (Imbeau et al. 1999), as well as salvage logging following fire disturbances (Hoyt and Hannon 2002, Morissette et al. 2002). Many studies suggest that suitable areas of postfire forests should be maintained intact, that is, not become subject to commercial salvage logging (Beschta et al. 1995, Hutto 1995, Saab and Dudley 1998, Hobson and Schieck 1999, Hoyt 2000, McIver and Starr 2000, Morissette et al. 2002, Stepnisky 2003). However, it is still unclear how much burned area should be considered suitable to be left unsalvaged.

This study presented the interactions of harvesting, fire, and salvage logging under a stochastic fire regime, and simulated the effects of different salvage logging alternatives and scenarios on timber harvested levels and habitat areas in the long term. On the basis of the defined threshold values of salvage logging, certain burned habitat areas were kept in reserve to promote maintenance of biodiversity in burned stands. The results were provided in the form of quantile distributions of timber harvest levels, NPVs, and habitat areas in both productive and passive landbases. This model is offered as a means to represent the probabilities of outcomes when a highly variable component in forested ecosystems, i.e. the annual burn rate, is factored into the projections. Previous research on salvage logging focused either on the ecological values or on the timber values of burned stands. This study incorporated both, and provided cumulative results over 200 years that arose under different salvaging scenarios. Taking into consideration landscape composition and the forest management plan priorities within a forest area, forest managers can derive a salvage threshold by reviewing the tradeoffs between timber harvest levels and sustaining habitat areas. The balance of fires and salvage logging works to effect losses of either habitat areas or timber volumes; in the long term, however, they may critically affect the richness and abundance of woodpeckers, and greatly impact the financial revenues of forest companies as well. In Scenario 3 with the assumptions of reforestation in the salvaged stands and natural regeneration in the unsalvaged stands, tradeoffs show that the median of the harvest levels in the FMA area is not greatly affected by increasing the threshold in the long term. A certain amount of burned habitat areas would be reserved in this scenario. However, particular large fires would create an opportunity for forest companies to obtain more timber volumes through salvage of burned trees with lower timber dues and salvaged volumes not chargeable against AAC. Therefore, an appropriate salvage logging strategy should ensure that forest companies recover the losses from fires, while upholding the requirements for sustaining habitat areas.

Timber harvest levels and NPVs differed among the different salvage logging alternatives and treatment scenarios in this study. Very few studies, when measuring or questioning the health and sustainability of forest ecosystems, have factored salvage logging strategies into forest management planning. Without an active salvage plan, the burned habitat for certain wildlife would be at peril from conventional salvage logging practices. The cumulative effects from salvage logging would make forest managers reconsider and react on fire lands, in particular, to reduce impacts on the biodiversity of forest ecosystems. Recognition of fire's highly variable characteristics should help forest managers and the public understand the relative risks of salvage logging alternatives and silvicultural treatments in the burned stands.

The study also computed burned habitat areas for two woodpecker species in the passive landbase over 200 years (most areas are non-merchantable forests), where it was assumed that no salvage logging occurred. Few studies have demonstrated whether non-merchantable postfire forests serve identical functions for woodpeckers as merchantable ones, or how much of burned area is required for the persistence of their populations. If the burned stands in the passive landbase could provide a sufficient area for woodpeckers, a different salvage logging strategy would direct forest companies to manage burned stands in the productive landbase. Unfortunately, no such information is available, and this remains outside of the scope of the present study.

The present study places greater emphasis on timber harvest levels than on NPVs when analyzing the influences of salvage logging on forest company practices. Timber conversion returns fluctuate from time to time and from place to place. It is difficult to forecast real timber prices or even short-term costs into the future with any accuracy. Therefore, in view of the inherent uncertainty associated with assessing the profitability of salvage logging strategies for forest companies, timber harvest levels serve as the better basis for projecting values from a forest management planning perspective. Nevertheless, NPVs could still be used to analyze the estimated financial benefits from salvage logging alternatives at current conditions, and so provide perspectives for future-directed management strategies.

The fire simulation in this study emulates the conditions associated with stand-replacement wildfires, assuming that fire suppression is kept constant. Fire severity is also an important aspect, one that is worth exploring in a future study. In fact, fire severity does influence both the quality of resultant habitat and the degree of bark retention, both of which affect woodpeckers' ability to reproduce and feed (Greenberg 1995, Hoyt and Hannon 2002, Stepnisky 2003). Information generated from this model could be applied to a general understanding of the effects of salvage logging in burned forests over the long term. Although I have discussed only two species of wildlife characteristically abundant in burned areas, there are many other species with preferences for various stages of fire-created habitat in the boreal forest that could be evaluated under the projections of burned areas by cover type and postfire habitat stages.

2.6 Bibliography

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Appendix 1: The code for timber supply in Woodstock programming

```
ACTIONS
*ACTION harv Y Harvest
*OPERABLE harv
  ? ? ? n  wood >= 50

AREAS
*A    c  sw    G   n
   1 250  806  35  286  398   653  1404  3322  3233  4808  5267  7262 2230 8888 1231  521  85  26  3  15
...
...
CONTROL
*LENGTH   20
*GRAPHICS ON
*REPORTS  ON
*SCHEDULE ON

LANDSCAPE
*THEME landbase
  c  conifer landbase
  d  deciduous landbase
  cd conifer dominated mixedwood
  dc deciduous dominated mixedwood
*AGGREGATE conland
  c cd
*AGGREGATE decland
  d dc
*THEME Cover type
  sw   White spruce
  sb   Black Spruce
  pi   Pine
  mc   conifer dominated mixedwood
  md   deciduous dominated Mixedwood
  dd   deciduous
  lt   larch stand
*AGGREGATE con
 sw sb pi mc lt
*AGGREGATE dec
  md dd
*AGGREGATE mx
  md mc
*THEME TPR
  G    Good
  M    Medium
  F    Fair
  U    Unproductive
*AGGREGATE prod
  G M F
*THEME deletion
  n    no buffer no deletion
  b    waterbody buffers
  d    subjective deletion
```

LIFESPAN
```
? ? ? ? 40
```

OPTIMIZE
```
*OBJECTIVE
_MAX totvolhar 1.._LENGTH
*CONSTRAINTS
 _EVEN(totconhar) 1.._LENGTH
 _EVEN(totdechar) 1.._LENGTH
 _NDY(stockconifc) 11.._LENGTH
 _NDY(stockdecid) 11.._LENGTH
*FORMAT XA
```

OUTPUTS
```
*OUTPUT areacut_dec deciduous area cut
*SOURCE decland ? ? ? harv _AREA
*OUTPUT areacut_con conifers area cut
*SOURCE conland ? ? ? harv _AREA
*OUTPUT areacut_tot total area cut
*SOURCE areacut_con + areacut_dec
*OUTPUT totconhar total conifer timber cut
*SOURCE ? con ? ? harv wood5
*OUTPUT totdechar total deciduous timber cut
*SOURCE ? dec ? ? harv wood5
*OUTPUT totvolhar total timber harvest
*SOURCE totconhar + totdechar
*OUTPUT stockconifc total commercial conifer stock
*SOURCE conland ? prod n _INVENT wood
*OUTPUT stockdecid total conifer growing stock
*SOURCE decland ? prod n _INVENT wood
```

TRANSITIONS
```
*CASE _DEATH
 *SOURCE ? ? ? ?
 *TARGET ? ? ? ? 100
*CASE harv
 *SOURCE ? ? ? ?
 *TARGET ? ? ? ? 100
```

YIELDS
```
*Y ? sw G ?
wood 1 0 0 0 22 91 156 212 262 305 341 371 397 420 438 455 479 483 494 494 494
*Y ? sw M ?
wood 1 0 0 0 0 2 50 96 140 179 215 247 276 301 324 343 362 378 393 393 393
*Y ? sw F ?
wood 1 0 0 0 0 0 0 0 0 13 43 71 98 124 149 172 194 213 233 250 250 250
*Y ? sw U ?
wood 1 0 0 0 0 0 0 0 0 0 0 13 22 31 39 47 47 47 47 47 47
129 170 207 239 268 293 316 336 354 370 385 398 398 398
...
...
*YC ? ? prod ?
wood5 wood * 0.9
```

Chapter 3.

A simple spatial fire model

3.1 Introduction

In the boreal forest of Alberta, wildfire is one of the major natural disturbances (Lee 1999, Song 2002). Large fires in particular have important effects on forest ecosystems and vegetation dynamics across a landscape over time (Johnson 1992). Increasingly, forest operations are attempting to emulate natural disturbances across the landscape in order to have ecologically sound harvest activities (Hunter 1993, Hutto 1995, Stelfox 1995, AFCSSC 1997, OMNR 2001). Hunter (1993) described the natural disturbance model (NDM), which suggests that timber harvesting can emulate the disturbances associated with wildfire. He identified three ways in which forest practices could mimic forest fires: the frequency, the size and pattern, and the amount of residual materials remaining. An approach for emulating natural fires over a large extent landscape requires knowledge of fire disturbance patterns, including their size and configuration. To this end, a spatial fire model, which could generate considerably detailed spatial projections of both fires and remnant patches, would prove of great value in determining the characteristics of fires' spatial spread, as well as for predicting both their ecological effects and their influences on forest production capacity in the long term.

In Chapter 2, salvage logging after fires had been analyzed non-spatially. But fires and logging activities are spatially relevant – both affect forest structures, the dynamics of vegetation, and wildlife habitat. In order to present those spatial characteristics of fires, a certain type of spatial fire model is required. It should be capable of modeling and simulating a long-term fire regime in order to better provide the simulation of the fire configuration overall, and thus to provide results that can become efficiently incorporated into forest management planning (Turner et al. 1994, Green et al. 1995, Gardner et al. 1999, Hargrove et al. 2000).

Of models that simulate fire growth or spread, there are some that are based on fire's chemical, physical, or even geometrical principles. These models require detailed information on local weather conditions (temperature, wind direction, wind speed, humidity, etc.), landscape topography (slope, aspect, and elevation), and of vegetation structure and characteristics of local fuels (species, and foliage moisture content) (Van Wagner 1977, Von Niessen and Blumen 1988, Hargrove et al. 2000, Miller and Yool 2002). These fire models are designed to predict localized fire spread over limited time scales (Hargrove et al. 2000) by analyzing a fire's thermodynamics and chemistry, and by modeling the parameters associated with fire behavior's prediction and control (Clarke et al. 1994). Still other models are based on cellular automata (CA) theory, which uses the fixed distances between neighboring cells, and involves the discrete process of burning through a grid-based landscape (Clarke et al. 1994, Berjak and Hearne 2002).

Cellular Automata (CA), which are "mathematical models for complex natural systems containing large numbers of simple components with local interactions" (Wolfram 1984), have been successfully used in modeling fire spread. Using the properties of simple structures while having the capability of exhibiting complex dynamic processes, CA provides an alternative to traditional equation-based models, and can be used to describe many physical systems and processes (Wolfram 1984, Karafyllidis and Thanailakis 1997). A CA model is constructed from a finite number of regular uniform cells, which are joined together to form a lattice. Square cells are usually used in a lattice. The state of each cell is updated at discrete time intervals subject to predefined rules based on its own states and on the states of its neighbors (Wolfram 1984). CA techniques provide the framework for integrating available spatial data and are appropriate for modeling fire growth and spread (Berjak and Hearne 2002, Karafyllidis and Thanailakis 1997). Spatial fire models have been developed to describe and predict the sizes and the locations of potential fire occurrences. Current spatial technologies have increased our ability to perform the integration, manipulation, analysis, and display of large complex sets of spatial data, and in this way provide a means for the visualization of wildland fire spreading.

Probabilistic models of fire spread, which are constructed on the basis of the CA technique and use estimated probabilities of fire start and spread, are regarded as one of the most efficient simulators (Gardner et al. 1999, Hargrove et al. 2000). Probabilistic models are usually used to characterize the heterogeneity of forest conditions and to predict final patterns of burns at a broad extent. A probabilistic model studied by Hargrove et al. (2000) simulated fire patterns based on percolation theory by incorporating various estimated spread probabilities in a heterogonous landscape, taking different characteristics of four successional stages of lodgepole pine (*Pinus contorta*) forests into consideration. Percolation theory is used to predict random processes that fill cells on a lattice and form clusters of connected cells (Kesten 1982). The size of a cluster is closely related to the percolation critical threshold, which is defined from the outset. Percolation theory has been used to simulate forest fire spread. However, the formulation of percolation results in a type of static model, since, in order to form fire clusters (Caldarelli et al. 2001) and limit the largest fire size, a percolation critical threshold must be pre-defined. It is possibly of more uses in simulating one fire event rather than a natural fire regime. Few studies focus on either the analysis of fire size distribution in the long term or on the fire patterns generated within percolation fire models.

Most spatial fire models are square-cell based (for example, Turner et al. 1994, Clarke et al. 1994, Cumming et al. 1998, Andison and Marshall 1999, Hargrove et al. 2000, etc.). Squares share edges with 4 neighbors but also touch another 4 with only one point. Therefore, the number of neighboring cells can be either 4 or 8, depending on the definition of a model. When a fire spreads to the defined 4 neighboring cells from an ignited one, the fire's growth pattern is in some manner stepwise because the fire can only go toward neighbors in 4 directions. When a fire spreads to 8 neighbors, the drawbacks for modeling lie in the unequal distances that a fire spreads out from each cell to its 8 neighbors, and in their unequal numbers of the neighboring cells of burning blocks in diagonal directions, which causes unequal likelihood of unburned neighboring cells being ignited by burning ones (Hargrove et al. 2000). Therefore, the fewer number of neighboring cells on diagonals than on axes results in less probability of their being ignited. This often complicates the fire spread along diagonal directions, and the unequal distances among neighbors may influence the pattern of a fire's spread.

The hexagon-based model is potentially superior to the grid cell, because hexagons touch all six neighbors along shared equal edges, and the centre of each neighbor is equidistant. Frandsen and Andrews (1979) first simulated line fire behavior in heterogeneous fuel beds that were depicted as hexagonal cells. Rempel et al. (1999) used hexagonal cells to represent the landscape and map the landscape patterns created by historical fire disturbances, thus providing insight into how to emulate the patterns of natural disturbances in the harvesting cutblocks.

PATCH (Schumaker 1998) is a spatially explicit wildlife habitat simulator which predicts the effects of land use changes on populations of terrestrial vertebrate species. PATCH presents the landscape as hexagons to minimize spatial distortion and provides an equal distance for animal movement from cell to cell. The apparent benefit of hexagonal representation has led to this approach being used in many analogous applications. This study will similarly present a fire-spread model using the hexagonal cell lattice.

Fire spread is controlled by factors including forest fuel, climate, and topography. These factors interact and influence fire ignition and spread, and consequently affect a fire's size and pattern. Including all of them into a simple model and predicting a fire's effects at the landscape level is unwieldy, due to highly variable elements such as weather change and the dynamics of vegetation succession. However, as CA probabilistic models are concerned only with the final size and pattern of a burn, consideration gets restricted to the fire contagion process from one cell to another, i.e. to the start and stop time points. Fuel, which represents the organic material available for fire ignition and burning, is a critical element in fire behavior, and the spatial distribution of fuel significantly influences the pattern of fire spread. Wind is another critical factor affecting fire spread and burn pattern. However, it is not easy to include wind in a fire model since wind varies by the minute, and is altered by topography and even by the fire itself (Gardner et al. 1999). Wind changes its speed and direction and strongly influences a fire's spread rate, direction, and size. Even small changes can produce dramatic difference in a fire's spread. Variations in topography influence the fuel moisture and the rate of fire spread; however, they vary over space but are constant over time.

The objectives of this study were to: 1) build a spatial fire model based on CA techniques, 2) examine whether a simple CA model can generate fires similar to natural ones over large spatial areas and across long temporal passages, and 3) examine whether a hexagonal fire model might better simulate fire patterns than a square-based model in terms of fire size distributions and fire configurations. The simulations were carried out by depicting a landscape as an array of hexagonal cells, and then building a simple fire model that incorporated fuel types and weather factors in a large landscape. Fires generated from models were compared with actual fires in terms of fire shape, size, and annual burn areas. Fire islands associated with fire size were also examined. A Monte Carlo simulation was applied in this study to randomly choose the location of fire ignition, the wind speed and direction, and the likelihood of fire spreading from cell to cell. The fire size was controlled via the spread probability of each fuel type. Owing to the stochastic nature of the spread, any fires – even ones started in the same place twice – would always be different. The study also examined the minimum number of parameters that might be required in the fire simulation by statistically analyzing the results from different simulation scenarios. The model simulation was performed using the Matlab program (Math Works 2001) and GIS techniques.

Cumming (1997, 2001*a*, 2001*b*) and Cumming et al. (1998) have studied fire behavior and spread in the boreal forest Alberta, and presented a spatial fire model, which was square-cell based with a 3 ha resolution, used in forest management and ecological processes. This study differed from Cumming's (1998) and others in the following aspects:

1. The heterogonous landscape was represented as an array of hexagonal cells instead of square ones.
2. Fire spread probabilities as related to fuel types were computed from annual burn rates.
3. Fire sizes were controlled using fire spread probabilities which were calibrated by using actual fire data instead of the predefined critical threshold to restrict fire sizes.
4. The wind direction and speed were randomly assigned based on historical data, and spread probabilities were adjusted during the fire simulation process.
5. The distribution of individual burned areas and annual burn areas were simulated.
6. Fire pattern and unburned island information were generated.

3.2 Study areas and actual fire data

The roughly rectangular study region, bounded approximately by 53° 30'N and 110° W, 58° N and 116° W, was chosen to examine the fire history (Figure 3.1). The total area of approximately 11 million ha consists mostly of boreal mixedwood forest (Strong 1992). The topography is relatively flat.

Alberta Sustainable Resources Development (SRD) provides an online historical fire database from 1961 to 2000 (SRD 2003), which includes the location of fire ignition, final fire size, wind speed and direction at the time of initial fire attack, and other useful information. From this database, 1,191 fires greater than 3 ha that originated in the study region were extracted. Wind data associated with these fires were summarized by providing percentages of 9 groups described in terms of wind direction: no wind, north, northeast, east, southeast, south, southwest, west, and northwest winds, and further categorized into 3 groups in terms of wind speed categories (weak winds with speeds less than 3 km/h, moderate winds with speeds between 4 and 22 km/h, and strong winds with speeds greater than 23 km/h (Hargrove et al.)). Table 3.1 shows the percentages of wind direction and speed, which were used as sources for the probabilities of wind directions and speeds randomly appearing in the model's simulation.

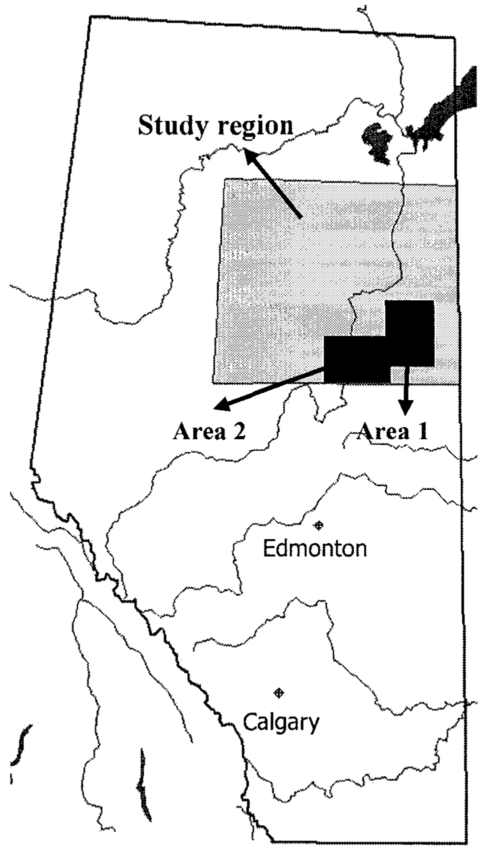


Figure 3.1. Location of study areas and the fire region where the fire history was examined

Table 3.1. Historical wind information on the percentages of wind direction and wind velocity in the study region.

Wind direction	Percentage of wind direction (%)	Percentage of wind velocity (%)			Total
		Weak (≤3km/h)	Moderate (4-22km/h)	Strong (≥23km/h)	
No wind	7				
North	4	5	93	2	100
Northeast	6	5	92	3	100
East	5	2	93	5	100
Southeast	15	6	81	12	100
South	6	10	83	7	100
Southwest	24	5	82	14	100
West	18	4	82	14	100
Northwest	15	1	89	10	100
Total	100				

Areas 1 and 2 in Figure 3.1 were used to develop the spatial fire model and calibrate the model's parameters. They are approximately 0.7 million ha each, accounting for 6.4% of the total study region. The landscape was classified into two types: flammable and nonflammable. The flammable type was subdivided into five forest fuel types: aspen (*Populus tremuloides*), white spruce (*Picea glauca*), black spruce (*Picea mariana*), jack pine (*Pinus banksiana*), and other, based on the primary species in the Alberta Vegetation Inventory (AVI) database (AEP 1991). The type "other" included non-forested areas, predominantly wetlands, and shrubs. Balsam fir was considered to be equivalent to white spruce, and larch to black spruce, for the purpose of this study. The nonflammable type included water and bare soil areas where no fire ignited. Table 3.2 shows the annual number of fires and annual burned areas from 1961 to 2000 in areas 1 and 2; these were used to analyze and compare the annual burned areas derived from the model's simulations.

The SRD online database also provides geographic information system (GIS) coverage of the boundaries of all fires greater than 200 ha during the period 1960-1997 and greater than 12 ha during 1998-2000. I redefined the projection of the fire coverage as UTM and Zone 12, in order to keep the fire and forest inventory maps in the same geographic area. Two hundred nineteen fire polygons, including those initiated inside but burned out of the boundary of the study region were extracted.

Most large fires contain unburned islands. Eberhart and Woodard (1987) digitized and examined 69 Alberta boreal fires ranging in size from 21 to 17,770 ha, and analyzed both the fires' size and shape, and the number and size of islands of residual vegetation. Their study provided detailed information on real fire patterns. Although the study had some deviations in fire size and shape from the SRD owing to the precision in the determination of the burned boundary (Eberhart and Woodard 1987), both were used to validate the fire patterns simulated from fire models in this study.

Table 3.2. The annual number of fires and burned areas in study areas 1 and 2 from 1960 to 2000.

	Area 1		Area 2	
Year	Number	Area (ha)	Number	Area (ha)
1961	1	57.06	6	970.65
1962	0	0	0	0
1963	0	4.04	1	3.64
1964	4	490.85	3	731.06
1965	0	0	0	0
1966	1	80.93	0	0
1967	2	174.41	0	0
1968	1	3.23	1	6.07
1969	1	8.09	0	0
1970	6	552.83	0	0
1971	4	372.11	3	222.97
1972	6	1,984.99	5	705.31
1973	0	0	0	0
1974	1	8.09	0	0
1975	1	3.23	0	0
1976	1	80.93	3	27.50
1977	0	0	1	12.14
1978	0	0	4	2,224.93
1979	1	4.04	10	7,636.02
1980	7	2,452.00	3	1,988.89
1981	5	152.95	3	4,269.25
1982	4	1,210.68	14	32,655.43
1983	1	23.60	1	3.60
1984	2	124.10	2	36.00
1985	2	1130	0	0
1986	3	26.60	0	0
1987	5	64.90	2	24.50
1988	3	314.10	3	31.00
1989	4	140.50	1	34.80
1990	10	1,275.20	4	21,846.90
1991	7	2,343.40	3	290.70
1992	3	14.50	1	3.00
1993	5	123.50	2	248.30
1994	5	114.00	3	69.50
1995	10	1,275.20	2	174.60
1996	3	17.00	0	0
1997	2	30.00	0	0
1998	8	136.50	16	18,369.90
1999	18	13,720.00	13	11,196.90
2000	3	86.50	4	670.20

3.3 Development of the hexagonal fire model

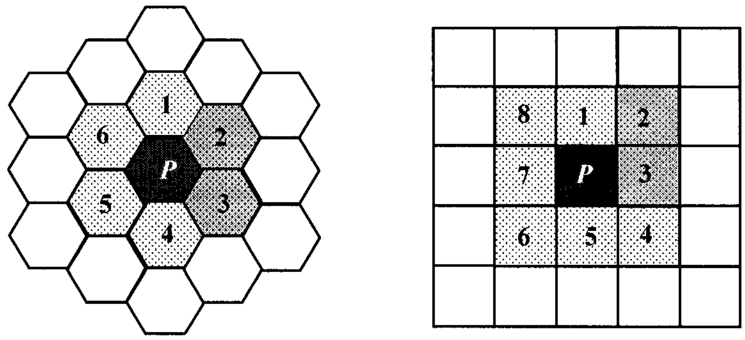
3.3.1 Basic principles of fire growth and spread

The hexagonal-grid based fire model (called hexagonal model hereafter) in this study represented a landscape as an array of hexagons with a resolution of 3 ha, which is a convenient threshold value for analysis of fire size distribution (Cumming 2001*a*). Each cell is uniform in fuel type. Fire is a contagious process; the likelihood of a cell burning depends not only on its probability of doing so but also on the number of neighboring cells burning. Fire spread from each ignited cell to any of unburned neighbors was an independent event with a calculated spread probability P, where P ranges from 0 to 1, dependent on the fuel type. Fire spread involved three steps: (1) computing the P of each of the cells adjacent to a burning cell, to which the fire may spread, (2) generating a random number P_r, (3) comparing the P with the P_r and determining whether the fire can spread. This process was stochastic. A neighbor of a burning cell that did not burn may become a neighbor of another burning cell and possibly burn at a later time. Fire initiation can occur randomly across the landscape except in nonflammable areas, such as water bodies or bare soil lands. When modeled fires reached any boundaries of the study area, a toroidal shift (Upton and Fingleton 1985) was applied in order to remove the effects of that fires' dispersing would be stopped beyond the boundaries with no fuel. The toroidal shift (Upton and Fingleton 1985) configures the rectangular or square study area of as a torus, where the top edge of the study area is conceived as abutting the bottom edge of the area, and left edge as abutting the right edge. The purpose is to keep intact the sequence and spatial or temporal structure of the data (Fortin and Jacquez 2000). Therefore, fires spreading out of boundaries propagate continually at the opposite edges. These fires can be considered as ones that are initiated inside the study area but which may burn outside of it. The burned area could be as small as one cell or larger than thousands of cells.

The probability of a cell's being ignited when a neighboring cell is burning depends on what type of fuel it is comprised of and on the number of neighboring cells that are burning (Figure 3.2 a). The probability of a cell's ignition is:

$$P = 1 - \prod_{i=1}^{n} (1 - P_s) \qquad [3.1]$$

where P_s is the spread probability of the cell that is potentially ignited by burning neighbors. P_s is related to its fuel type. n is the number of neighbors that are burning. P is the probability of being ignited for the cell. If there are no burning neighbors adjacent to the cell, the probability of its being ignited is 0.



a b

A potentially ignitable cell
Neighboring cells, 1 to 6 for a hexagon, 1 to 8 for a square
Burning neighboring cells

Figure 3.2. Depiction of the principle of fire spread.

3.3.2 Fire spread probabilities

Fire ignition and spread are fuel-dependent. However, few studies have been done on fire spread probability in terms of fuel types because of the difficulties involved in obtaining basic spread probabilities related to fuel types and owing to lack of fire maps detailing the before and after burning areas, such as would be needed to compare and assess a model's accuracy. Thus, to test fire models against historical data would require many years of observations. Some probabilistic fire simulations employed identical fire spread probability (for example, Clarke et al. 1994), assuming a homogeneous landscape, whereas some employed estimated ones associated with fuels (for example, Hargrove et al. 2000). Cumming (2001*b*) analyzed the relationship between wildfire behavior and fuel types in the Alberta boreal mixedwood forest based on the historical fire data and presented the annual burn rates for the main tree species. Cumming (2001*b*) found that deciduous stands burned at the lowest rate, and black spruce stands burned at the highest rate. In this study, I assumed that the burn rate of each fuel type (Table 3.3) may reflect its flammability and can be used to represent its spread probability.

Table 3.3. Annual burn rate for each fuel type (Cumming 2001*b*)

Fuel type	Aspen	White spruce	Black spruce	Pine	Other
Burn rate (%)	0.05	0.17	0.50	0.42	0.17

3.3.3 Wind effects

Wildfires are strongly influenced by wind (Hirsch 1996, Hargrove et al. 2000, Berjak and Hearne 2002). Fire spread probability is sensitive to variations in wind speed and direction. Wind is generally viewed as contributing to a fire's traveling from a flaming area to an area of unburned fuel downwind, and as transforming the shape of a fire from a circle, in the case of no wind, to an ellipse (Berjak and Hearne 2002). In this study, wind effects were simulated by assigning a larger spread probability in the downwind direction. Once ignited, a fire could spread randomly to an adjacent cell but was more likely to move toward the downwind cells. Wind direction was randomly determined at a fire start. Hargrove et al. (2000) grouped wind speeds into three classes and applied a bias factor *b* to adjust the spread probability for each neighboring cell. I used their idea of wind bias factor and applied it in the model, using various wind directions, rather than a fixed one. The model assumed that weak winds with speeds ranging from 0 to 3 km/h had no effects on the fire spread. For moderate winds with speeds between 4 and 22 km/h and strong winds with speeds greater than 23 km/h, the bias factor and a cumulative binomial equation (Hargrove et al. 2000) were used to adjust fire spread probabilities of downwind neighboring cells (Equation 3.2), in accordance with the wind direction. With no wind or weak winds, the value of bias factor was 1, so the spread probability did not change; moderate and strong winds increased the possibility of fire spreading in downwind directions by values of 2 and 3 respectively. Equation 3.2 shows how to adjust the probability by assigning the bias factor value:

$$P_{wi} = 1 - (1 - P_{si})^{bi} \qquad [3.2]$$

where i indexes the neighboring cells. P_w is the spread probability of a cell after being adjusted; P_s is the spread probability of a cell before being adjusted, and *b* is the wind bias factor. *b* can hold the value of 1, 1.5, 2, 2.5, or 3 depending on the wind speed and direction, as well as the position of the downwind cells and their adjacent cells.

Figure 3.3 shows examples of how wind bias values were assigned to the downwind cells and their adjacent cells when west or north winds exist during the cells' burning. Since wind strongly influences fire growth and spread, larger fires occur in extremely windy conditions. For example, if a west wind with a moderate speed (4-22 km/h) blows, the eastern neighboring cells are twice as likely to be ignited by the burning cell. In this case, the fire spread probability in a downwind aspen stand is: $P_w = 1 - (1 - 0.05)^2 = 0.098$, and in a downwind black spruce stand $P_w = 1 - (1 - 0.50)^2 = 0.75$.

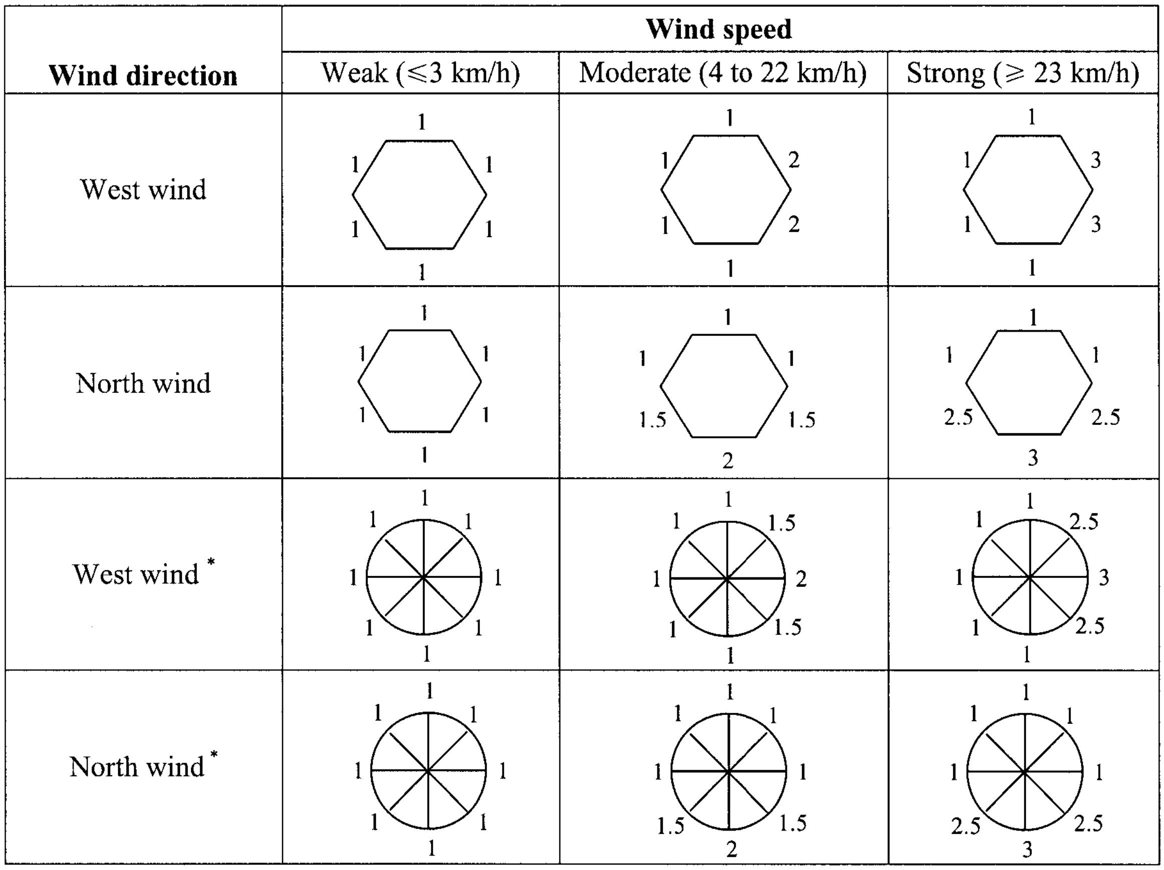


Figure 3.3. Examples of wind bias factor b assigned to a west and a north wind (*Hargrove et al. 2000).

3.4 Model verification

Model verification was conducted using existing land inventories. Two areas were selected to test the fire simulation model (Figure 3.1). Each area was depicted as an array of hexagonal cells, and each cell represented an area of 3 ha and was assumed to be homogeneous. The model propagated a fire by defining a set of probabilities of a fire spreading from cell to cell. The probabilities depended on the fuel type, and they were adjusted for each cell, depending on wind speed and direction during the simulations. Models were calibrated using three scenarios by incorporating different fire factors. In each scenario, 1,191 fires were generated in each simulation and analyzed in terms of the fire size distribution.

3.4.1 Scenario 1 – simulation in homogeneous forests

With identical fire spread probabilities for all fuel types in flammable areas, a small probability P_s=0.10 was chosen to start the fire spread simulation. The model also tested that when P was less than 0.10, no fires larger than 40 ha were generated, and when P was greater than 0.50, there were too many large fires, some of which could even burn the entire study areas. Therefore, the range of P ($0.1 \leqslant P \leqslant 0.5$) was applied in the model. By increasing P with 0.01 in each simulation, the model presented the changes in the fire size distribution. Four simulation results as examples were shown in the result and represented the changes in fire sizes with different fire spread probabilities. As nonflammable areas did not contribute fires, P is 0. This scenario was an attempt to examine the basic model assuming the homogeneous forests and to evaluate whether the model would produce fires that matched the actual fires.

3.4.2 Scenario 2 – simulation in heterogeneous forests

Fires burn unevenly. The rate and the pattern of fire spread are influenced by the spatial distribution of forest fuel types. Fires initiate at random cells and are more likely to spread to more flammable vegetation types. In Scenario 2, the annual burn rates in Table 3.3 were used to represent the spread probability of each fuel type. However, the modeled fires did not match historical fire data – more large fires occurred in the model, resulting in the modeled fire size distribution being far away from the actual fire distribution. I gradually decreased the annual burn rates by 2% for all fuel types in each simulation, and examined the changes in the distributions through extensive replications. Similarly, four sets of spread probabilities were listed as examples and demonstrated how the fire distribution changed with various spread probabilities.

3.4.3 Scenario 3 – simulation with wind factors incorporated based on Scenario 2

The direction of wind is the most likely direction for fire spread. The probabilities of fire spread depended on the forest fuel type, and they were adjusted according to the wind speed and direction, both of which were randomly distributed before a fire started during the simulation, according to the percentages of historical wind directions and speeds when fires occurred. Table 3.1 shows the percentages of no wind and 8 wind directions, as well 3 groups of wind velocities for each direction. During the simulation, the model generated 2 random numbers: one was used to determine the wind direction, and the other to determine the speed class. Thus, the wind bias values were assigned to the cells in downwind directions, and the spread probabilities were modified accordingly.

3.4.4 Simulation steps

For each fire, the initiation and spread process involved the following steps:

Step 1. Randomly select one cell as the start point of a fire. If the fuel type of the cell is nonflammable, no fire starts. Reselect a cell until it is flammable, and a fire starts.

Step 2. For each neighboring cell of the randomly selected burning cell

i. Check whether it is flammable: if not, the cell will not be ignited.

ii. If flammable, calculate P, the probability of its being ignited, which depends on the spread probability (P_s) of each fuel type and on the number of burning neighbors.

a. P_s is identical in Scenario 1.

b. P_s is associated with the fuel type in Scenario 2.

c. P_s is adjusted by wind bias factor in Scenario 3 by

– Randomly choosing the wind direction and speed according to their probabilities effective when fires occurred, from historical wind information.

– Assigning the bias values to downwind cells.

iii. Randomly generate a number between 0 and 1 (P_r), compare P with P_r, and determine whether the cell is ignited.

iv. Record burning cells as burned ones.

v. Record new ignited cells as burning ones.

Step 3. Check whether the record of burning cells is null:

i. If yes, there are no more cells to be ignited and the fire stops.

ii. If not, list all neighbors of all burning cells.

iii. Repeat all items in Step 2.

Step 4. Record the fire size.

3.4.5 Shape metrics

One simulation, which generated fires with the best fit for fire size distribution, was used to analyze fire shape metrics. Fires larger than 12 ha were selected to analyze fire shape, and they were presented as the following fire size classes (Eberhart and Woodard 1987): class C (12 – 40 ha), class D (41 – 200 ha), class E_1 (201 – 400 ha), class E_2 (401 – 2,000 ha), and class E_3 (greater than 2,000 ha).

The FRAGSTATS package (McGarigal and Marks 1995) uses several statistical inputs to quantify landscape configuration according to the complexity of patch shape. The perimeter and area ratio (*PAR*), the shape index (*SI*), and the patch fractal dimension (*PFD*) were used to examine the irregularity and complexity of fire shapes in this study. They are calculated as:

$$PAR = \frac{P}{A} \qquad [3.3]$$

$$SI = \frac{P}{2 \times \sqrt{\pi \times A}} \qquad [3.4]$$

$$PFD = \frac{2 \ln P}{\ln A} \qquad [3.5]$$

where P is the length of the perimeter (m) and A is the area (m^2) of a fire.

The *PAR* is a simple measure of the shape complexity. A limitation of this index is that it varies with the size of patches even though the shapes of the patches are same. The *SI* overcomes the size problem of the *PAR* as a measure of the shape complexity by adjusting for the circular standard (polygons). The value of *SI* equals 1.0 when a fire is a perfect circle and increases without limit as the fire's shape becomes more irregular. The *SI* could be the simplest and perhaps most straightforward measure of overall shape complexity. The *PFD,* a measure of the boundary complexity, is derived from the perimeter and area relation. The value of *PFD* is between 1 and 2; it approaches 1 for shapes with simple perimeters and approaches 2 when shapes are more complex, with highly convoluted, plane-filling perimeters (McGarigal and Marks 1995).

The existing spatial fire coverage shows fire polygons only for fires greater than 12 ha, and does not provide any data on unburned fire islands. In the fine scale, for example 3 ha, more ragged boundaries of these real fires affect the values of the shape matrices, compared with the modeled fires that have relative straight edges within the resolution. A modeled fire was a cluster of hexagons with skips (fire islands) inside boundaries. In order to make real fires comparable with modeled fires, real fire polygons were converted to the patterns with hexagonal boundaries of 3 ha resolution, and all skips in the modeled fires were filled (Figure 3.4).

Fire shape is an important criterion to describe fire spread pattern and its potential effects on the landscape. Fire shape metrics of actual fires were compared with those of simulated fires from Scenario 3, and with the study of Eberhart and Woodard (1987), which provided the information on *SI* and fire islands. The Mann-Whitney test was used to decide whether, subject to different fire classes, but not to individual fires, significant differences emerged between the fire shapes of actual fires, of the simulated fires of the hexagonal and square models, and of Eberhart and Woodard (1987).

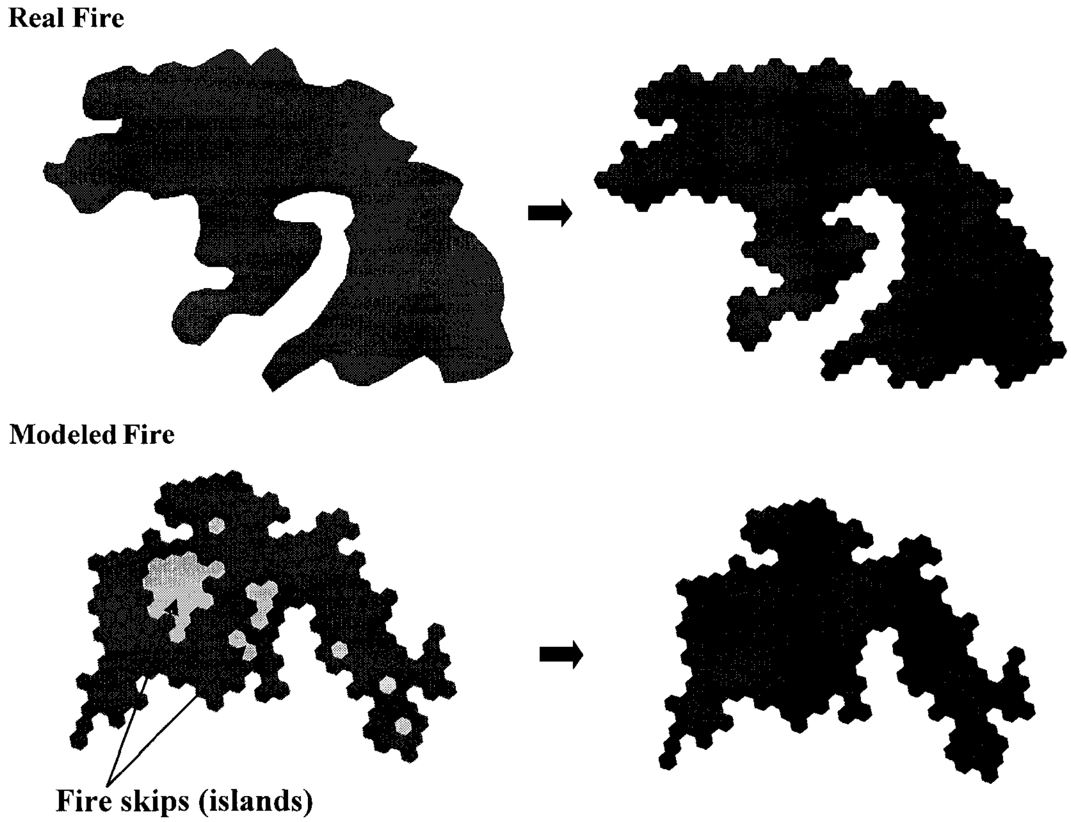


Figure 3.4. The boundary of a real fire was converted into a hexagonal boundary pattern, and all fire skips in a modeled fire were filled.

3.4.6 Comparison with the square-based fire model

The same procedures for developing the hexagonal model were used to generate a square-based fire spread model (called square model hereafter) in order to compare whether these two models were similar in simulating fire events in terms of fire size, annual burned area, and fire shape. The square model viewed the number of neighboring cells in a central square as 8, a difference from the hexagonal model. Thus the probability of each cell catching fire from its burning neighbors was different as well (Figure 3.2 b). Consequently, the spread probability of each fuel type needed to be modified accordingly. The equations used to compute shape metrics needed to be adjusted as follows to be applicable to the square model (McGarigal and Marks 1995):

$$SI = \frac{0.25 \times P}{\sqrt{A}} \quad [3.6]$$

$$PFD = \frac{2\ln(0.25 \times P)}{\ln A} \quad [3.7]$$

Fire size distributions were compared and tested using the Kolmogorov-Smirnov (K-S) test and the Mann-Whitney (M-W) U-test, in order to reveal the goodness of fit between simulated results and historical data. Both tests are applied for non-parametric datasets. The K-S test tries to determine whether two samples are distributed identically based on the difference between the relative cumulative frequency distributions of the two samples (Sokal and Rohlf 1995). The M-W test is used to determine whether two samples have the same location, such as the median and quantiles. It is used to compare entire probability distributions based on the sums of the ranks and measure differences in location of the two samples (Sokal and Rohlf 1995). The K-S test examines differences in dispersion, shape, and location of the distributions and thus is the more comprehensive test, but it is less powerful with regard to the location of two samples than the M-W test (Sokal and Rohlf 1995). Both tests were used to evaluate the differences between the modeled and actual fires. If the results from the two tests match, conclusions can be drawn with greater confidence. Statistical analyses of the data were performed using SPSS for Windows (SPSS Inc. 2002). A significance level $p \leqslant 0.05$ was used for all statistical analyses in this study.

3.5 Results and discussion

3.5.1 Fire size distribution

Tables 3.4 and 3.5 show four sets of fire spread probabilities which were applied in scenarios 1, 2, and 3 for both hexagonal and square models. These four probabilities covered most ranges from small to large probabilities and presented the levels of fitness to the actual fire data. Figures 3.5 and 3.6 present the cumulative distributions of fire sizes from the historical fire database and from the simulation results. In Scenario 1, where the identical fire spread probability was used in homogenous forests, none of the simulations with different spread probabilities was able to fit the actual fire size distribution (p value < 0.001). Scenario 2 simulated the condition of the fire spread based on the spread probability of each fuel type. With many simulations in which different sets of spread probabilities were modified, results showed that although the fire size distributions were closer to the actual fire data than in Scenario 1, they were still statistically different from the distribution of actual fires (p value < 0.001).

Table 3.4. Fire spread probabilities in scenarios 1, 2, and 3 applied in the hexagonal model

P	Scenario 1					Scenario 2					Scenario 3				
	Aspen	White spruce	Black spruce	Pine	Other	Aspen	White spruce	Black spruce	Pine	Other	Aspen	White spruce	Black spruce	Pine	Other
P_1	0.200	0.200	0.200	0.200	0.200	0.030	0.102	0.300	0.252	0.102	0.029	0.099	0.290	0.245	0.099
P_2	0.250	0.250	0.250	0.250	0.250	0.035	0.119	0.350	0.294	0.119	0.030	0.102	0.300	0.252	0.102
P_3	0.300	0.300	0.300	0.300	0.300	0.038	0.129	0.380	0.319	0.129	0.031	0.105	0.310	0.260	0.105
P_4	0.350	0.350	0.350	0.350	0.350	0.040	0.136	0.400	0.336	0.136	0.032	0.109	0.320	0.269	0.109

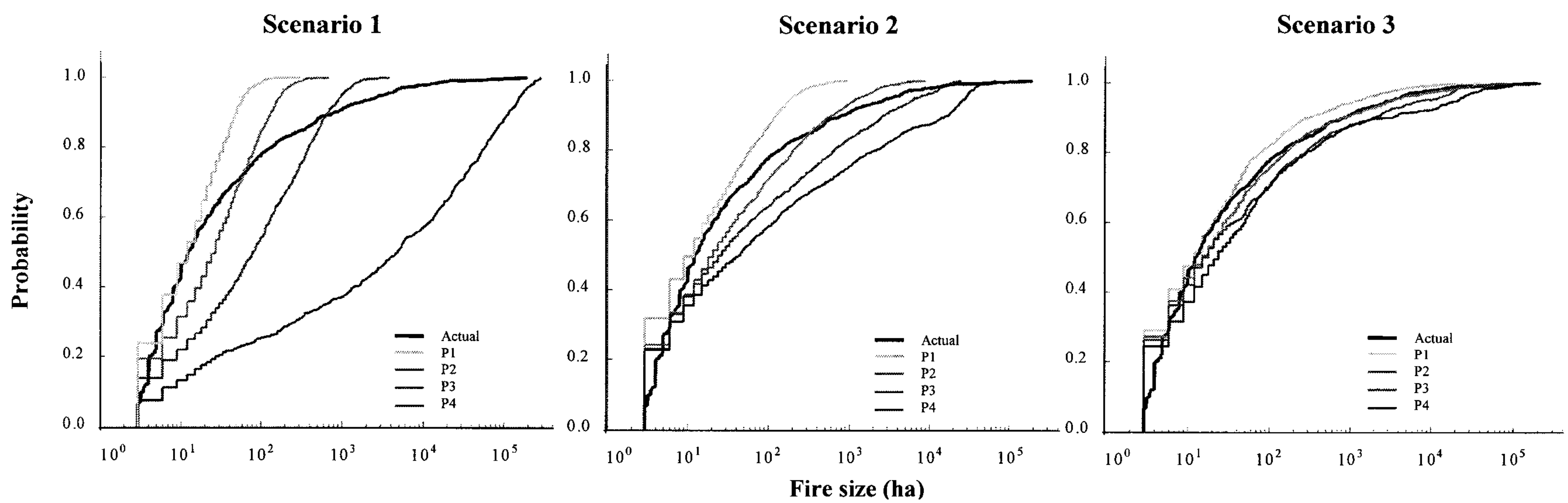


Figure 3.5. Cumulative distributions of fire sizes simulated in scenarios 1, 2, and 3 in the hexagonal model, compared with the actual fire data. Four simulation results, as examples, represent the changes in fire size distributions with the different fire spread probabilities (P_1 to P_4). Actual represents the actual fire data, and P_1, P_2, P_3, and P_4 in each scenario refer to Table 3.4. X-axis is a logarithmic scale.

Table 3.5. Fire spread probabilities in scenarios 1, 2, and 3 applied in the square model

	Scenario1					Scenario 2					Scenario 3				
P	Aspen	White spruce	Black spruce	Pine	Other	Aspen	White spruce	Black spruce	Pine	Other	Aspen	White spruce	Black spruce	Pine	Other
P_1	0.150	0.150	0.150	0.150	0.150	0.022	0.075	0.220	0.185	0.075	0.020	0.066	0.200	0.168	0.066
P_2	0.200	0.200	0.200	0.200	0.200	0.025	0.085	0.250	0.210	0.085	0.021	0.071	0.210	0.176	0.071
P_3	0.220	0.220	0.220	0.220	0.220	0.028	0.095	0.280	0.235	0.095	0.022	0.075	0.220	0.185	0.075
P_4	0.250	0.250	0.250	0.250	0.250	0.030	0.102	0.300	0.252	0.102	0.023	0.078	0.230	0.193	0.078

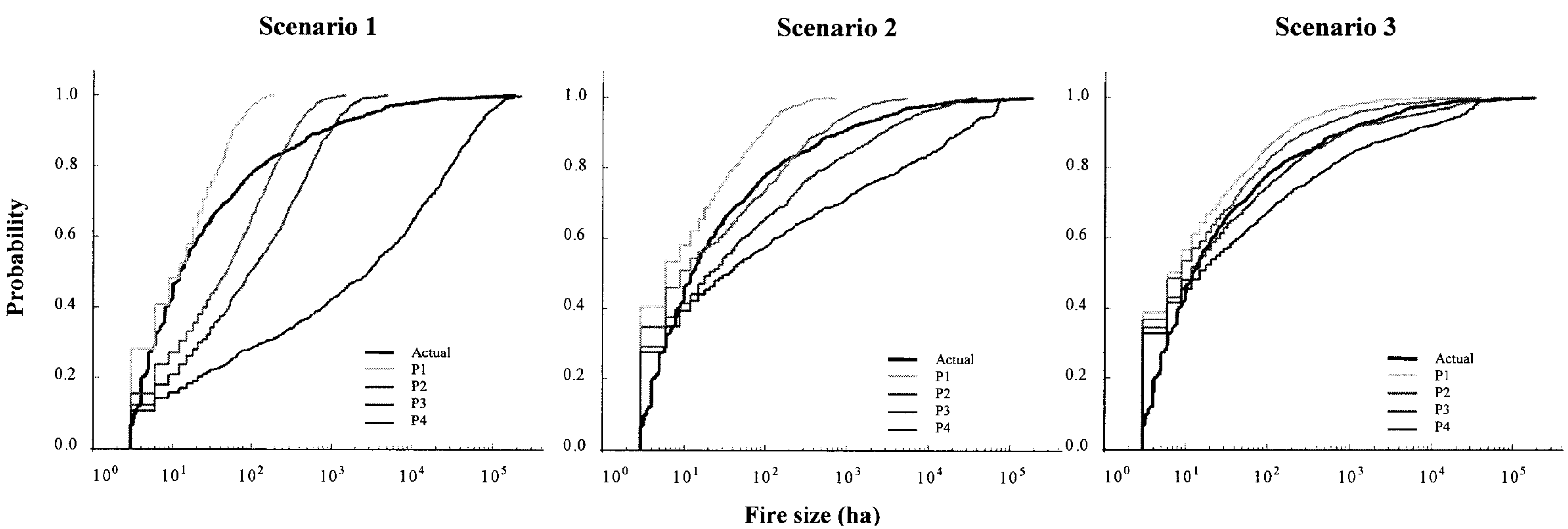


Figure 3.6. Cumulative distributions of fire sizes simulated in scenarios 1, 2, and 3 in the square model, compared with the actual fire data. Four simulation results, as examples, represent the changes in fire size distributions with the different fire spread probabilities (P_1 to P_4). Actual represents the actual fire data, and P_1, P_2, P_3, and P_4 in each scenario refer to Table 3.4. X-axis is a logarithmic scale.

Fire spread under the influence of wind revealed rather different results. Scenario 3 took wind speed and direction into account and adjusted spread probabilities by the bias factor. One simulation with the probabilities in Table 3.6 yielded a close visual fit for the actual fire sizes, but statistical tests showed significant differences between them (p value < 0.001). However, after examining the distribution of fires greater than 12 ha, Scenario 3 produced the modeled fires best overall fit with actual fire size distribution (p value > 0.05). This is because as fires burned across one cell to another, the fire size jumped from 3 ha to 6, to 9 ha, etc. Figures 3.5 and 3.6 also show the step-like incremental pattern as results linked to small fires moved from 3 ha to 6 or 9 ha, thus affecting the overall tests of goodness of fit. Since small fires less than 12 ha contributed negligibly to the total area burned (< 0.2%, Table 3.7), those fires were ignored in the analysis of the fire size distribution.

Table 3.6. Spread probabilities best fit for the actual fire data.

	Hexagonal model					Square model				
	Aspen	White spruce	Black spruce	Pine	Other	Aspen	White spruce	Black spruce	Pine	Other
P	0.030	0.102	0.300	0.252	0.102	0.022	0.075	0.220	0.185	0.075

After calibrating the set of the spread probabilities that represented the simulated fires in study area 1 fit for the actual fire data, this set of values was applied to area 2 to further test whether it was applicable in different forest areas. Simulation results showed that there were minor differences in fire size distributions between areas 1 and 2, but both were considered as fitting actual fires (p value > 0.05). All the above analyses were generalized for both the hexagonal and square models. Table 3.7 lists the comparisons of fire sizes simulated in areas 1 and 2, as well the results of statistical tests. Figure 3.7 summarizes the simulations of Scenario 3 in areas 1 and 2 for both hexagonal and square models, in a manner that gives a better representation of fire class distributions. The set of spread probabilities in Table 3.6, which was the best fit for the actual fire data in Scenario 3, was selected to conduct analyses of annual burned areas and fire shape metrics in the following sections.

Table 3.7. Comparisons of fire size distributions for Scenario 3. Values in bold indicate no significant differences between modeled fires and actual fires.

Fire size (ha)		Actual fires	Hexagonal model		Square model	
			Area 1	Area 2	Area 1	Area 2
Mean		1,319	1,106	1,257	1,309	1,995
Median		12.2	12	15	12	18
Max		187,008	166,428	156,948	90,795	135,018
Percentage of fires < 12 ha (%)		0.2	0.2	0.2	0.2	0.1
p-value for all fires	K-S test	N/A	< 0.001	< 0.001	< 0.001	< 0.001
	M-W test	N/A	< 0.001	< 0.001	< 0.001	< 0.001
p-value for fires ≥ 12 ha	K-S test	N/A	**0.415**	**0.312**	**0.234**	**0.07**
	M-W test	N/A	**0.906**	**0.189**	**0.677**	**0.07**

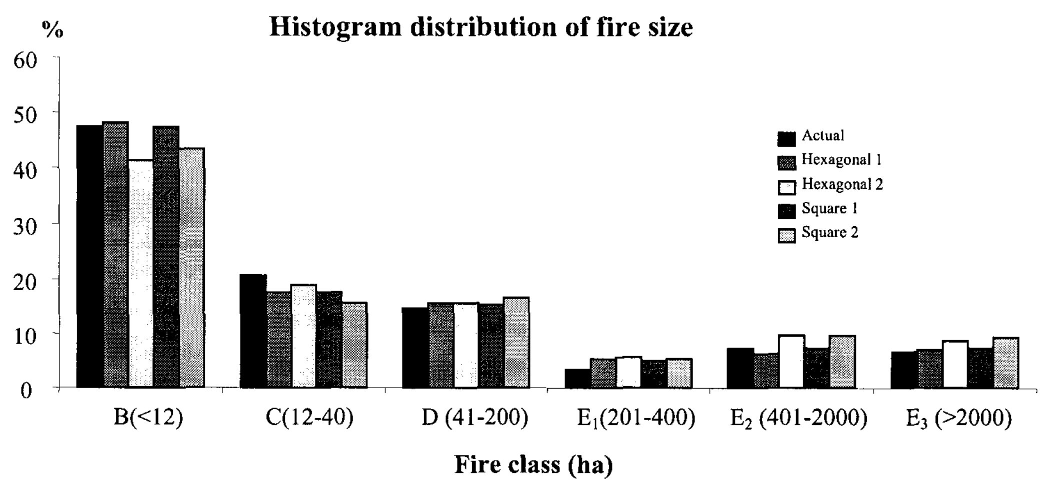


Figure 3.7. Histogram distributions of fire classes simulated in scenario 3. 'Actual' indicates the actual fires. 'Hexagonal 1' and 'Hexagonal 2' represent the fires simulated from the hexagonal model in areas 1 and 2, and 'Square 1' and 'Square 2' from the square model in areas 1 and 2.

3.5.2 Annual burned areas

The simulations of annual burned areas were conducted in both study areas 1 and 2. The number of fires occurring in a year was randomly selected from the historical annual fire numbers in each study area for a 40-year period simulation (Table 3.2). The 40-year period is the duration of the available fire history. Simulations were replicated 100 times. Figures 3.8 and 3.9 present the results of annual burned areas from each run for a 40-year period as the cumulative distributions ranging from the lowest quantile of 0.025 and the highest quantile of 0.975. The results revealed that 95% distribution intervals of the simulated annual burned areas from both hexagonal and square models cover most of the actual fires in areas 1 and 2. Both models simulated annual burned areas well.

3.5.3 Shape metrics

The results of the shape metrics are shown in Figure 3.10, representing the comparison between the modeled fires, actual fires with converted boundaries, and study of Eberhart and Woodard (1987). With the increase in fire size, the value of *PAR* becomes smaller, and *SI* and *PFD* become bigger. This indicates that the shape of larger fires is more irregular and complex, and the fire perimeter more convoluted. When comparing the fires simulated in the hexagonal model with actual fires that were converted to hexagonal boundary patterns, the modeled fire shape was similar to that of actual fires in classes C and D, and became more similar to that in the study of Eberhart and Woodard (1987) in classes E_1, E_2, and E_3. Differences in the starting fire size – 12 ha in both actual and simulated fires, but 21 ha with a non-converted boundary in the study of Eberhart and Woodard – might have underlined the variations in the small-fire shape index. In the other case, where fires simulated in the square model were compared either with actual fires that were converted to square-boundary patterns or with the study of Eberhart and Woodard, the shape metrics showed differences in *PAR, SI,* and *PFD.* Table 3.8 summarizes the results of comparisons. The values of *PFD* for hexagonal fires (modeled and actual) in Figure 3.10 are greater than those for square fires due to the differences in configuration between the hexagonal lattice with ragged edges and the square lattice with straight edges.

3.5.4 Fire islands

The occurrence of fire islands or fire skips, which are the areas undisturbed or lightly disturbed within a fire boundary, is common and accounts for a substantial portion of large fire events (Andison 2003). Simulated fires in class C did not involve any unburned islands, coinciding with

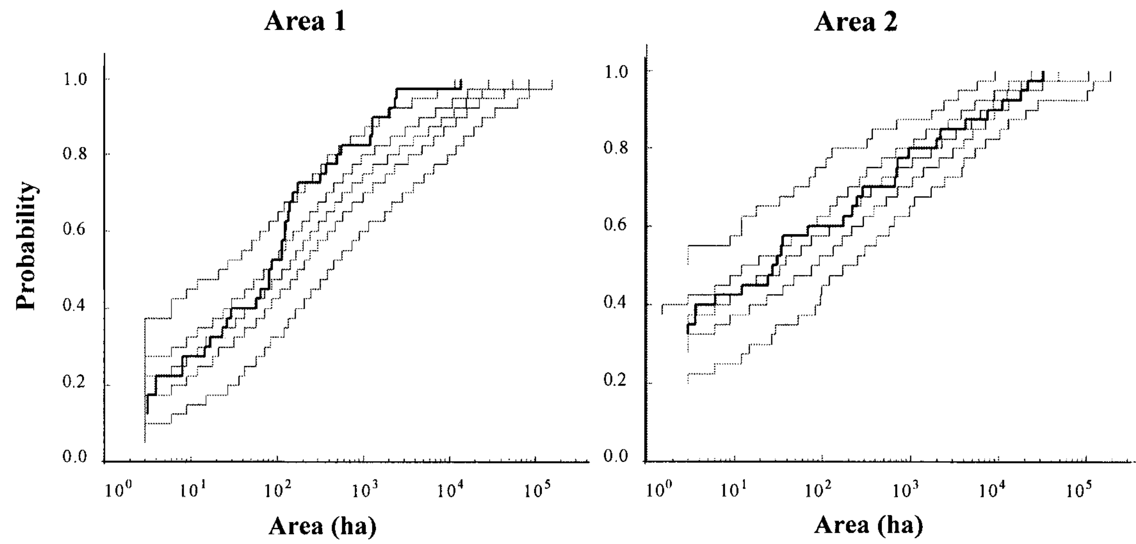


Figure 3.8. Cumulative distributions of annual burned areas in areas 1 and 2 simulated in the hexagonal model. The dark lines represent actual fires, and the thin lines are the quantile distributions of the simulated fires. From bottom to top, the quantiles are 0.025, 0.25, 0.50, 0.75, and 0.975.

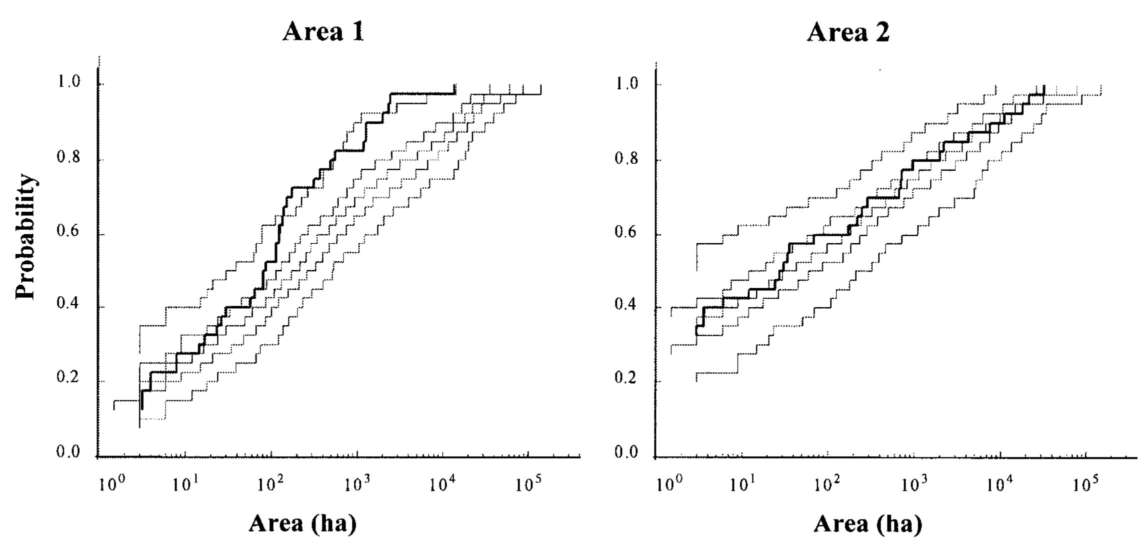


Figure 3.9. Cumulative distributions of annual burned areas in areas 1 and 2 simulated in the square model. The dark line represents actual fires, and the thin lines are the quantile distributions of modeled fires. From bottom to top, the quantiles are 0.025, 0.25, 0.50, 0.75, and 0.975.

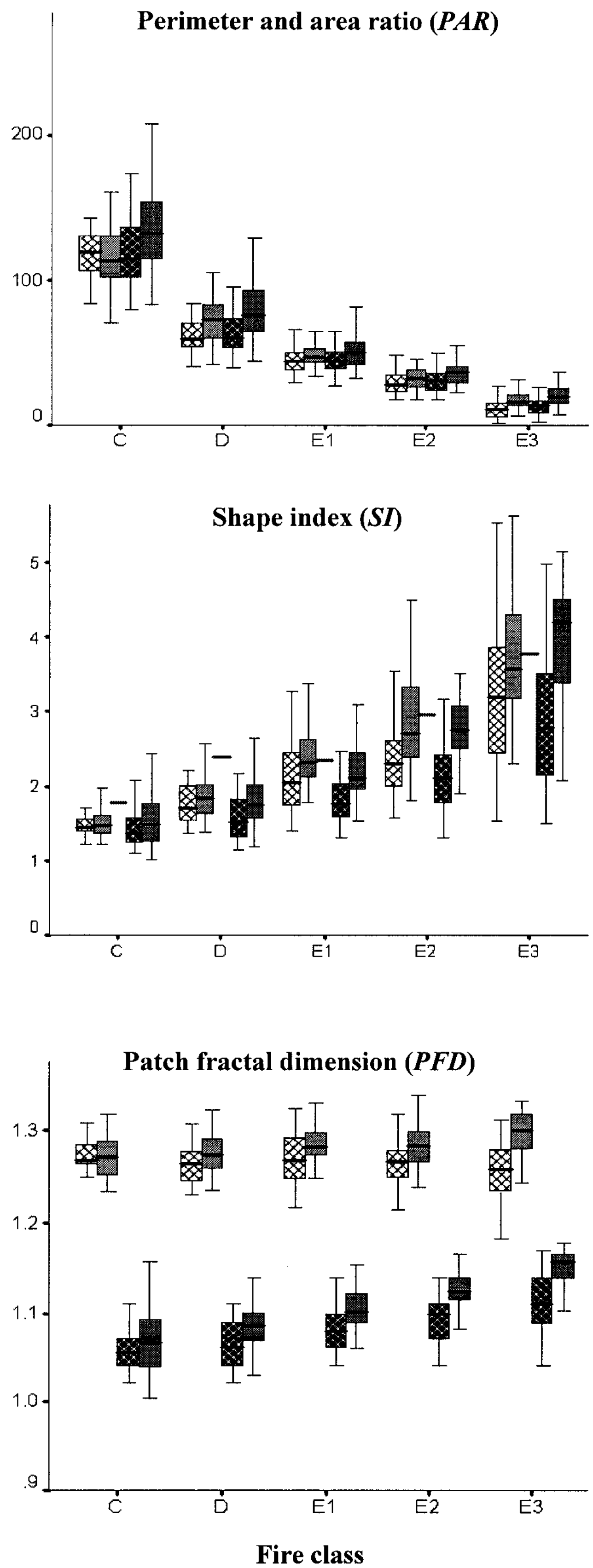


Figure 3.10. Boxplots of *PAR*, *SI*, and *PFD*, compared actual fires with both hexagonal and square models and with the study of Eberhart and Woodard (1987). The length of box is the interquartile range with a median bar inside. The top and bottom bars are the maximum and the minimum.

Table 3.8. The number of fires greater than 12 ha obtained from actual fires, from Eberhart and Woodard (1987), and from hexagonal and square model simulations; medians of *SI*, *PAR* and *PFD*; the number of islands per 100 ha; and median island areas by fire size class. The Mann-Whitney test was used to test the significant differences. 'Actual_H' and 'Actual_S' refer to the real fires converted to hex-boundary and square-boundary patterns. 'Hexagonal' and 'Square' refer to the simulation results from hexagonal and square models. 'Eberhart and Woodard' refers to the results from their study (1987).

		C (12-40)	**D (41-200)**	**E_1 (201-400)**	**E_2 (401-2000)**	**E_3 (>2000)**	**Total**	**Max fire size (ha)**
Number of fires	Actual	21	28	32	76	62	219	187,011
	Eberhart and Woodard	8	20	13	16	12	69	17,770
	Hexagonal	177	157	44	52	63	493	174,123
	Square	157	173	47	91	39	507	90,759
Median of *SI*	Actual_H	1.455	1.716	2.051	2.311	3.199		
	Actual_S	1.375	1.53	1.785	2.12	2.79		
	Eberhart and Woodard	1.79	2.4	2.36	2.96	3.78		
	Hexagonal	**1.485**[1]	**1.854**[1]	***2.321***[2]	***2.718***[2]	***3.574***[2]		
	Square	1.5	1.772	2.126	2.76	4.2		
Median of *PAR*	Actual_H	119.4	60.35	45.35	28.7	11.55		
	Actual_S	115.47	61.51	44.83	30.96	13.74		
	Hexagonal	**114.6**[1]	74	**47.45**[1]	32.8	16.4		
	Square	133.2	77	50.4	37	20.1		
Median of *PFD*	Actual_H	1.268	1.264	1.268	1.2665	1.2575		
	Actual_S	1.055	1.064	1.08	1.1	1.112		
	Hexagonal	**1.271**[1]	**1.273**[1]	1.2815	1.285	1.3		
	Square	1.066	1.086	1.1016	1.125	1.153		
Number of islands/100 ha	Eberhart and Woodard	0	0.38	0.96	0.87	0.39		
	Hexagonal	0	0.21	0.73	0.85	1.05		
	Square	0	0.88	1.23	1.16	1.42		
Median size of islands	Eberhart and Woodard	0	2.29	2.5	2.59	9.39		
	Hexagonal	0	3	3	3	3		
	Square	0	3	3	3	3		

[1] Numbers in bold indicate there is no significant difference between modeled and actual fires

[2] Numbers in italic bold indicate there is no significant difference between modeled fires and Eberhart and Woodard's (1987)

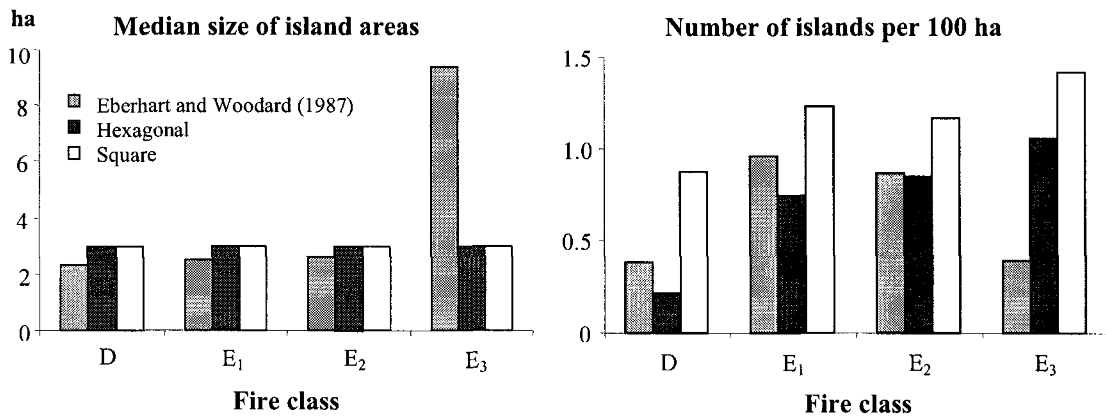


Figure 3.11. Comparisons of median island areas and the number of fire islands per 100 ha simulated from hexagonal and square models, and from Eberhart and Woodard (1987).

results from the study of Eberhart and Woodard (1987). Therefore, islands in fire class C were excluded from this analysis.

Comparing simulated fires from both models with those reported by Eberhart and Woodard (1987), two models generated the same median size of island areas, which was similar to the study of Eberhart and Woodard, except for the fires in class E_3. Both models produced more fire islands per 100 ha with the increase in the fire size, different from the study of Eberhart and Woodard, which showed fewer in the larger fires. The hexagonal model generated the number of islands per 100 ha slightly better than the square model in fire class D, E_1, and E_2. Eberhart and Woodard showed fewer fire islands per 100 ha but larger median island areas in class E_3 (Figure 3.11, Table 3.8). Reasons for these differences may be traceable to the high intensity or long duration of the burning of a real larger fire, which in turn may allow it to reburn the escaped islands with changes in wind direction. However, in the model simulations, wind speed and direction remained constant from the start to the end of a fire. The square model generated more fire islands per 100 ha than both hexagonal model and the study of Eberhart and Woodard. The less probabilities of diagonal cells' being ignited result in more single cells being kept unburned, thus influencing fire burned sizes and island patterns.

3.6 Conclusions

This study presented a simple probability-based fire model which represented landscapes as hexagonal cells with a 3 ha resolution. The primary benefit of such a model is the possibility, by the advantages of its hexagonal spread context, to readily take into account the probabilities of fire spread associated with both the fuel types and wind conditions. The promise of the model is its possibilities for the simulation of fire a large extent landscape, over a long time frame, both feasible and simple.

Fuel and wind factors are important in determining the fire size distribution. Both the hexagonal and square models demonstrated well the distribution of individual fire sizes and annual burned areas. However, the hexagonal model simulated the fire pattern more natural than the square model, particularly with respect to the shape of large fires. This may have been due to the hexagon's characteristics – the equidistance to any of 6 neighboring cells from a central cell and the greater constancy in the probabilities regarding neighboring cells catching fire. This work showed that spread probabilities adapted from annual burn rates can better represent natural fire characteristics, and that using burn rates associated with fuel types was superior to using the identical spread probabilities in the fire simulations across a landscape. Wind velocity and direction strongly influence fire spread probability and burned size. Therefore, fuel and wind factors in fire simulations cannot be ignored in attempts to simulate fire size distribution and pattern.

Identifying areas that have a high ignition and burning probability is a critical component in fire models which are to be incorporated in forest fire management planning. This model can be used to simulate long-term fire effects in a large landscape and provide fire burn pattern and indicators of unburned islands within the large burns. Fire islands vary in number and size, and they perform ecological functions as habitat, cover, shelter, and seed source, and contribute as well to the diversity of forest age structure. An understanding of fire islands could provide knowledge of natural disturbance regimes for ecological forest management. Representing a landscape as an array of hexagonal territories is being used increasingly in analyze wildlife populations or timber harvests (Schumaker 1998, Calkin et al. 2002). If the fire regime model is to be incorporated into forest management practices and use made of long-term simulations, then this study will become beneficial for forest planning strategies, evaluating risks and allocating fire suppression resources, and in particular, to advance the emulation of natural disturbance patterns in the forest practices. In such a case, the effects of salvage logging as discussed in Chapter 2, could be simulated spatially.

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Chapter 4.

Conclusion

Two studies have been presented in this thesis both related to natural fire regimes and forest management concerns. In Chapter 2, the interacting disturbances of forest harvest, wildfire, and postfire salvage logging were simulated. The objective of this first study was to evaluate appropriate rates of salvage logging and to develop tradeoff curves correlating timber harvest levels and wildlife habitat areas under different salvage scenarios. Different salvage logging areas, (i.e. salvaging all burned stands, salvaging none of the burned stands, salvaging with thresholds of 2,000, 5,000, 10,000, and 20,000 ha) and subsequent silvicultural treatments (reforestation in the burned stands after salvaging, no reforestation in the burned stands, and assumed natural regeneration in the unsalvaged stands) were proposed and simulated. Monte Carlo simulations of forest fires and an optimization-based forest harvesting model were used to project annual allowable cuts, net present values, and habitat areas over a 200-year planning horizon, and the simulation was replicated 1000 times. Probability distributions of projected outcomes were generated, presenting the different levels of habitat areas and harvest levels, subject to different salvage strategies. The results showed that a tradeoff might be able to be effected between the two management goals of maintaining certain burned areas and maximizing the timber supply. These results could be used to help forest managers decide an appropriate salvage rate.

Forest management has shifted from purely managing timber products towards sustainable forest management concerned with maintaining biodiversity and enhancing ecosystem integrity (AFCSSC 1997, Davis et al. 2001). Natural disturbance regimes are encouraging forest harvest practices to emulate natural patterns. However, harvested blocks are not analogous to ones created by fires (Lee 1999, Song 2002), and salvage logging alters the characteristics of burned stands structurally and functionally, placing it at odds with the natural disturbance regime. In order to better manage post-fire forests in Alberta, forest managers need to improve their understanding of how salvage logging can affect both timber harvest levels in the long term as well as the postfire habitat areas specific to some certain wildlife. A number of studies have examined fire effects on the landscape. However, the ecological value and function of standing burned trees has largely been ignored in salvage operations. As a result, salvage logging strategies tend to resemble clearcutting. In Chapter 2, various salvage logging alternatives and silvicultural scenarios were presented, and the consequences of salvage logging after fires were considered from the perspective of ecological forest management. Forest managers might use this or a similar model to make salvage decisions in light of priorities set by their forest management planning systems, and to project the timber supply, financial benefits, and ecological constraints that forest fires likely provide. This study, providing various streams of information on salvage logging, might contribute to salvage logging management methods, for example, by developing techniques to determine salvage volumes by finding salvage thresholds to put into effect within snag habitat areas. Another possibility would be to foster cost-benefit analyses of management options that include ecological and economic variables.

In Chapter 3, a simple spatial fire model was developed. The fire model used the cellular automaton approach to simulate the spatial spread of wildfire at large spatial and temporal extents. The model presented the landscape as an array of hexagons, and the spread probability of a fire was modified in accordance with annual burn rates of the main species in the study area. Wind speed and direction were incorporated by assigning a bias factor and adjusting the spread probability, in accordance with historical data on the wind speed and direction of real fires. Fire size distributions, fire patterns, unburned islands, and annual burned areas were simulated for a 40-year period. The hexagonal- and square-grid based models were compared, and the results showed that both hexagonal and square fire models could simulate fire size distributions well, but the hexagon based model better simulated the fire shapes and unburned islands in the study area.

Chapter 3 has presented methods for simulating fire spread by utilizing a cellular automaton (CA) approach and presenting fire size distribution and configuration within a large landscape. This model took advantage of hexagonal characteristics and incorporated CA and GIS techniques to simulate a natural fire regime at large spatial and temporal extents. The fire model took into account fuel types and wind factors that greatly influence the fire size and patterns in the study areas. With minimum factors as the input, the model could perform efficiently to simulate not only a random individual fire event, but also fire distribution for a given time period. The model could be put to use in such fields as wildlife habitat preservation, landscape dynamics, and forest production affected by fires. The original intention in building the fire model was to simulate salvage logging spatially. Salvage operations commonly occur in merchantable stands and readily accessible regions. When we implement the spatial fire model, more information than only the fire spatial location and size in the landscape arises from the simulation. For example, the total burned areas that can be accessed could be estimated by defining the distance between existing roads and the boundaries of burned locations. As these spatial-related data input, the salvage logging model may provide not only non-spatial analyses for forest management objectives, but also spatial dynamics for ecological concerns. The fire model could reveal the areas which are susceptible to fires, and map and model the areas that are at fire risks.

There is plenty of room for further research. For forest management to meet, as much as possible, both economic and ecological objectives, maintaining portions of burned landscapes from being salvaged will depend not only on the determination of thresholds, but also on the placements of the spatial locations of unsalvaged burned areas. Therefore, spatial simulation of salvage logging deserves to be further explored. Naturally-shaped blocks, residual clumps, variety of stand structures and compositions, and spatial arrangement should be incorporated into the management planning of the burned landscapes with multiple values. The ecological functions of the burned stands in the noncommercial landbase need to be clarified in order to identify a comprehensive salvage plan. The fire model in Chapter 3 did not take topography into account, as the study areas are relatively flat (Strong 1992). It is well known that the topography can be one of the critical factors influencing the fire spread. Variations in elevation and slope influence the fuel moisture and subsequently affect the flammability of fuel and the intensity of burn. However, these factors vary with space but are constant over time. The spatial fire model would be able to add one more attribute, i.e. the elevation in the landscape, when it is applied to other landscapes. Fire suppression efforts have been discussed in their roles of changing forest structure and altering fire regimes (Martell 2002, Ward et al. 2001). The effects of fire suppression on fire regimes would be worthy of further study.

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