**University of Alberta**

# Three Papers on the Natural Disturbance Model of Forest Management

by

**Glen William Armstrong** ©

A thesis submitted to the Faculty of Graduate Studies and Research in partial fulfillment of the requirements for the degree of **Doctor of Philosophy**.

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# Abstract

This thesis is a collection of three papers examining the natural disturbance model of forest management, specifically how timber harvest rates might be determined under that model. The studies presented here focus on areas of the boreal mixedwood forest in Alberta, Canada.

The first paper projects the consequences on timber supply of a simple interpretation of the natural disturbance model: the rate of disturbance (harvest and wildfire) under management is set equal to the rate of natural disturbance, and the proportion of different forest types subject to managed disturbance is equal to that under natural disturbance. The paper compares the timber supply implications of three very different estimates of the natural disturbance rate to the timber supply resulting from the current sustained yield policy. This interpretation of the natural disturbance model results in major reductions in annual allowable cuts.

The second paper characterizes the natural disturbance regime for an 8.6 million ha region of the boreal mixedwood forest as a serially independent random draw from a lognormal distribution. This characterization has some important results which were identified through Monte Carlo simulations. Estimates of the mean annual burn rate for the study area are highly variable. Single parameter characterizations of the disturbance regime (*e.g.* mean disturbance rate, fire cycle) are unreliable. There is no equilibrium age class structure for a forest subject to the lognormal disturbance regime. No target age class distribution can be justified on the basis that it is the "correct" natural distribution.

The third paper presents a modeling system that recognizes the variability inherent in the disturbance regime, and uses the variability to help guide harvest rate determination. Monte Carlo simulation is used to project the probability distributions of habitat areas for five vertebrate species for each year of the planning horizon. These probability distributions are used to set constraints for an optimization based forest planning model, which is used to develop trade-off curves between objective function values and constraint levels.

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Anja von Froesenhaus got me outside and away from my thesis on a regular basis. Christine Brodie is a wonderfully supportive friend, who also kept me away from my thesis at times. Without these distractions, this thesis would have taken much longer to complete.

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# Chapter 1

# Introduction

Forest management in much of Canada is in transition from multiple use, sustained yield management to sustainable forest management or, as some call it, ecosystem management. The goal of management is changing from the production of an optimum mix of commodities and services from the forest (*e.g.* timber, range, water, wildlife, and recreation) to the maintenance of the ecological integrity of the forest; the production of the commodities and services is assumed to be a byproduct of a healthy ecosystem. The Alberta Forest Legacy (Alberta Environmental Protection, undated) and the Canadian Council of Forest Ministers' Criteria and Indicators of Sustainable Forest Management (Canadian Council of Forest Ministers, 1997) are evidence of a political commitment to sustainable forest management in Canada.

Ecosystem health and ecological integrity are difficult concepts to define and measure. At the extreme, populations of hundreds of different species might need to be tracked: changes in these populations could provide indicators of ecosystem health and ecological integrity. However, this kind of monitoring is expensive and the complex relationships between ecosystem health and species populations are not well understood. Partly as a result of this difficulty, the natural disturbance model (NDM) of sustainable forest management has been developed (Hunter, 1993). The central hypothesis of the NDM is that management practices which emulate natural disturbances can be developed, and that the application of these practices will maintain ecological integrity. By mimicking natural patterns as closely as possible, NDM management is hypothesized to minimize the negative impacts of timber harvest on forest biota. This hypothesis allows the manager to focus on ecosystem process and structure, and how management techniques might emulate them, rather than on each species of concern.

Hunter (1993) identifies three ways in which timber harvesting practices can emulate natural disturbance: the frequency of harvest can be matched to that of natural disturbance, the size and distribution of harvest blocks can be matched to those of openings created by natural disturbance, and the amount of residual organic material left on-site after harvest can be matched to that of a natural disturbance. Hunter (1993) concentrated on the second of these points while Lee *et al.* (1997) considered the third. This thesis is concerned with the first of these points: the rate of natural disturbance.

Most of the discussion of NDM management for the boreal forests of Canada has centered on emulating the effects of stand-replacing wildfire. There are a number of possible reasons for this focus. Wildfire is the most important (or at least, most obvious) non-anthropogenic agent of change in the boreal forest (Johnson, 1992). To the extent that fire suppression efforts have been successful, timber harvest may be compensatory to wildfire. In other words, if fire suppression or other management practices have reduced the annual burn rate, timber harvesting may be used to make up the difference without changing the total area experiencing major disturbance each year. Some characteristics of a stand-replacing fire may be emulated by harvest techniques similar to those used in clearcut harvesting. Thus, extensive new investment in equipment and training may be unnecessary.

There is a continuum of spatial scales for which the effects of a wildfire may be apparent, ranging from the tree-level (or smaller) to the global level. For forest management purposes, two of the most important scales for analysis and prescription are the stand ($\sim$ 10 – 100 ha) and the forest ($\sim$ 100 000 – 10 000 000 ha). At the stand level, harvesting may be able to emulate some of the effects of fire. For example, some trees can be left unharvested to provide habitat structure in the regenerating stand (see, for example, Alberta Pacific Forest Industries Inc. (undated)). At the forest level, the distribution of stands by size class, their spatial relationships to each other, and the area and types of stands selected for harvest may be managed to emulate the patterns left by wildfire.

The determination of annual harvest area is possibly the most important forest level decision a forest manager can make. This decision determines how much timber the forest supplies annually and will, at least partially, determine the future age class structure of the forest: the decision has major financial and ecological implications. Hunter (1993) implies that the rate of timber harvest can be set so as to mimic the rate of natural disturbance in the boreal forest. This thesis presents the results of some explorations of the natural disturbance regime (NDR) and how an understanding of the NDR can be used to help guide harvest rate decisions.

This thesis is a collection of three papers, tied together with this introductory chapter and a concluding chapter. The first paper (Chapter 2, and a version published as Armstrong *et al.* (1999)) projects the consequences on timber supply of a simple interpretation of the natural disturbance model: the rate of disturbance (harvest and wildfire) under management should be set equal to the rate of natural disturbance, and the proportion of different forest types subject to managed disturbance should be equal to that under natural disturbance. The paper compares the timber supply implications of three very different estimates of the natural disturbance rate (0.5, 1.0, and 2.0%/year) to the timber supply that would result from implementation the current sustained yield policy.

The natural disturbance rates modeled in Chapter 2 were estimated by Murphy (1985) and Cumming (1997). They were developed for similar forests in northern Alberta using different methods. The magnitude of the difference between Cumming's and Murphy's estimates of the natural disturbance rate is important, because of the economic and ecological consequences of the harvest rate decision. The second paper in this thesis (Chapter 3 and a version published as Armstrong (1999)) began as an attempt to understand how the estimated rates of Cumming and Murphy could be so different, and to find out who, if anyone, was wrong. The annual area burned is characterized as a serially independent random draw from a lognormal distribution. This characterization is used to examine the variability of estimated mean disturbance rates and to try to characterize the equilibrium age class distribution.

Chapter 4 presents a modeling system that recognizes the variability inherent in the disturbance regime, and uses the variability to help guide harvest rate determination. The forest is characterized in terms of habitat area for selected vertebrate species. The forest is subjected to the natural disturbance regime presented in Chapter 3 for a 100-year planning horizon. This process is repeated 1000 times to generate a projection of the probability distributions of habitat areas for each year of the planning horizon. These probability distributions are used to set constraints for an optimization based forest planning model, which is used to develop trade-off curves between objective function values and constraint levels. The curves can be used by the owners and managers of the forest to decide what level of trade-offs are acceptable.

The concluding chapter summarizes the results, and presents some suggestions for further research.

# Bibliography

Alberta Environmental Protection. undated. The Alberta forest legacy: Implementation framework for sustainable forest management. Publication I-689, Alberta Environmental Protection, Edmonton.

Alberta Pacific Forest Industries Inc. undated. An Operator's Guide to Stand Structure. Alberta Pacific Forest Industries Inc.

Armstrong G.W. 1999. A stochastic characterisation of the natural disturbance regime of the boreal mixedwood forest with implications for sustainable forest management. Can. J. For. Res. **29**: 424–433.

Armstrong G.W., Cumming S.G., and Adamowicz W.L. 1999. Timber supply implications of natural disturbance management. For. Chron. **75**: 497–504.

Canadian Council of Forest Ministers. 1997. Criteria and indicators of sustainable forest management in Canada. Technical Report 1997, Canadian Council of Forest Ministers, Ottawa.

Cumming S.G. 1997. Landscape dynamics of the boreal mixedwood forest. Ph.D. dissertation, Department of Forest Sciences. University of British Columbia, Vancouver.

Hunter M.L. 1993. Natural fire regimes as spatial models for managing boreal forests. Biol. Conserv. **65**: 115–120.

Johnson E.A. 1992. Fire and Vegetation Dynamics: Studies from the North American Boreal Forest. Cambridge University Press.

Lee P.C., Crites S., Neitfeld M., Nguyen H.V., and Stelfox J.B. 1997. Characteristics and origins of deadwood material in aspen-dominated boreal forests. Ecological Applications **7**: 691–701.

Murphy P.J. 1985. Methods for evaluating the effects of forest fire management in Alberta. Ph.D. dissertation, Department of Forestry. University of British Columbia, Vancouver.

# Chapter 2

# Timber Supply Implications[1]

## 2.1 Introduction

One strategy for setting timber harvest levels under a natural disturbance model is to dictate that the average annual area disturbed under management should equal the average annual area disturbed under natural conditions. This presupposes that forest protection activities have the effect of reducing the annual area disturbed. The allowable annual harvest area is then the difference between the average area disturbed under natural conditions and the average area disturbed given the level of forest protection. In order to use this strategy, one must be able to determine average annual disturbance rates with and without fire suppression.

Alberta-Pacific Forest Industries Inc. (APFI) has proposed this strategy for determining annual harvest levels for their Forest Management Agreement Area in northeastern Alberta. They plan research to "determine the differences in disturbance rates across the FMA, so that harvest rates and methods can emulate the disturbance rate for each 'disturbance rate' zone" (Alberta Pacific Forest Industries Inc., undated). This approach to allowable cut determination is very different from the usual methods used in Alberta. In essence, the APFI approach assumes that the natural rate of disturbance can be determined for a forest area, and that this rate of disturbance should also be applied to the managed forest. In other words, the annual area disturbed (harvested and burned) under management should equal the annual area disturbed (burned) under natural conditions. To date, no examination has been made on the potential effects on timber supply which would result from the implementation of this approach.

Van Wagner (1983) presented a forest level simulation model incorporating a natural disturbance process similar to that modeled in this chapter, and timber harvest. The purpose of his model was to examine the effects of different annual rates of burn on the long term equilibrium timber supply of a forest. Reed and Errico (1986) examined a similar problem using an optimization framework. In both these papers, the objective was

[1]A version of this chapter has been published. Armstrong, G.W., Cumming, S.G., and Adamowicz, W.L. 1999. Timber supply implications of natural disturbance management. For. Chron. 75:497–502.

to determine the appropriate level of harvest given that a specified proportion of the forest was expected to burn each year.

Our model is less ambitious and serves a different purpose. We develop a simple simulation model of forest growth and disturbance. It is used to compare the areas and volumes disturbed under different estimates of the natural disturbance rate with results from a sustained timber yield model (without disturbance). The areas and volumes disturbed under the natural disturbance scenarios can be usefully thought of as defining the upper limits to areas and volumes harvested under the AFPI approach to natural disturbance management. The sustained yield scenario is intended to represent the *status quo* for harvest level determination in Alberta. I will show that the schedules of harvest area and volume for the different scenarios are strikingly different. The model presented here is an example of how a simple model can be a useful tool for evaluating the effects of a change in forest policy or practices. The results presented here are intended to provide a starting point for discussion of the relationship between the natural disturbance model of forest management and timber harvest levels.

## 2.2 Model Development

The model is a straightforward representation of forest growth and disturbance. The initial forest is represented as a matrix of productive area indexed by age and conifer content. Each year, a proportion of the area is disturbed and regenerates according to our regeneration assumptions. The portion of the area that is not disturbed increases in age by one year. A proportion of the area in each conifer content class moves to the next highest conifer content class. This sequence is repeated for each year in the specified simulation period. The inputs to the model used here consist of a set of forest inventory data, timber yield tables, succession and regeneration assumptions, and annual disturbance probabilities.

We obtained forest inventory data for a portion of the boreal forest in northeastern Alberta covering 300 townships (15 townships north by 20 townships east), or approximately $2.8 \times 10^6$ ha. The stand listings include a description of the forest cover type including species composition, stand age, and gross stand area as interpreted from aerial photographs. As part of a timber supply analysis for the area, the net productive land base of the study area was determined. Areas considered unmerchantable (including wetlands, swamps, and bogs), inoperable, or as part of reserves or buffers are excluded from the net productive land base. The net land base of the study area consists of 785 042 ha. I describe the forest in terms of the area in ten-year wide age classes and ten-percent wide conifer content classes. Conifer content classes are used here because stand inflammability is assumed to be related to conifer content, and they provide us with a way of tracking forest succession. The distribution of the inventory by age class and conifer content as of 1996 is is presented in detail in Table 2.1.

Forest growth is modeled through the application of softwood and hardwood yield tables. The yield tables used here were derived by Joy (1995) from the Alberta Phase 3 white spruce, mixedwood, and hardwood yield tables (Alberta Forest Service, 1985) calculated to the 13+/7 cm utilization standard. The softwood and hardwood yield tables used here are presented in Tables 2.2 and 2.3.

The succession model used here was developed by Cumming *et al.* (1995). They estimated a simple linear relationship between the photo-interpreted white spruce content of inventoried stands and the photo-interpreted age. Based on this relationship, the annual percentage increase in white spruce content of stands is estimated to be 0.37%. In the application used here, 3.7% of the area in each 10%–wide conifer content class moves to the next highest content class each year of the simulation.

Cumming (1997) also developed the regeneration model used here. The model assigns an age of zero to disturbed stands and allocates the area proportionately to conifer content classes using the proportions shown in Table 2.4. This empirical distribution was developed by examining the conifer content of stands originating in the 1960's and growing them backwards to age zero using the succession model described above. Using this model the probability of initial conifer content is independent of the conifer content of existing stands or management prescription.

I compare four different representations of disturbance regimes in this chapter. The sustained yield regime (scenario SY) is intended to represent sustained yield management as currently practiced in Alberta. Murphy (1985) estimates the average annual pre-suppression disturbance rate for northern Alberta to be approximately two percent: I represent this regime as scenario M2. Cumming (1997), using different methods, estimates conifer content specific disturbance rates for the region. His average annual disturbance rate is less than 0.5%: I represent this as scenario C1. For comparative purposes, I also develop a scenario using Murphy's model with a one percent annual disturbance rate (scenario M1).

As a basis for comparison, I refer to areas and volumes *disturbed* rather than *harvested* or *burnt*. The area disturbed in any particular year will be equal to the sum of the areas burnt and harvested, because fire suppression will never be completely effective. This relates directly back to the idea that in order for timber harvest to be compensatory to wildfire, fire suppression must be effective enough to reduce the disturbance rate.

Comparing the Murphy and Cumming regimes (M2, M1, and C1) to the sustained yield regime (SY) will provide some information on the timber supply implications of moving to a harvest scheduling system based on natural disturbance rates. Comparing the Murphy and Cumming regimes to each other may also be informative in that it provides a comparison of two independent estimates of the natural disturbance regime for the study area.

Table 2.1: Net area (ha) of starting inventory by age class and conifer content.

| | Conifer Content(%) | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Age(yrs) | 0–10 | 11-20 | 21–30 | 31–40 | 41–50 | 51–60 | 61–70 | 71–80 | 81–90 | 91–100 |
| 0–4 | 9 521 | 1 065 | 0 | 750 | 290 | 37 | 702 | 41 | 647 | 6 697 |
| 5–14 | 60 | 0 | 0 | 254 | 10 | 0 | 43 | 4 | 0 | 256 |
| 15–24 | 2 161 | 225 | 34 | 199 | 62 | 23 | 506 | 99 | 112 | 2 104 |
| 25–34 | 8 417 | 2 730 | 182 | 1 648 | 123 | 118 | 1 389 | 73 | 1 238 | 5 521 |
| 35–44 | 14 199 | 5 412 | 264 | 3 877 | 292 | 600 | 1 728 | 638 | 1 580 | 15 775 |
| 45–54 | 37 855 | 16 309 | 239 | 10 554 | 1 581 | 823 | 8 127 | 879 | 6 813 | 23 082 |
| 55–64 | 40 285 | 18 546 | 612 | 12 907 | 1 681 | 2 258 | 10 848 | 2 099 | 8 249 | 34 851 |
| 65–74 | 36 096 | 20 151 | 128 | 8 227 | 1 601 | 666 | 3 486 | 460 | 4 072 | 11 185 |
| 75–84 | 29 269 | 12 088 | 143 | 7 326 | 714 | 1 563 | 3 626 | 526 | 3 210 | 12 597 |
| 85–94 | 13 347 | 9 369 | 366 | 5 733 | 1 167 | 450 | 3 780 | 835 | 3 103 | 12 583 |
| 95–104 | 12 798 | 15 648 | 896 | 12 299 | 870 | 593 | 7 363 | 1 004 | 4 536 | 27 612 |
| 105–114 | 3 723 | 3 391 | 437 | 5 142 | 483 | 355 | 5 444 | 871 | 4 253 | 12 076 |
| 115–124 | 2 931 | 1 985 | 611 | 4 910 | 576 | 645 | 7 888 | 1 220 | 2 689 | 13 796 |
| 125–134 | 1 431 | 1 110 | 185 | 2 689 | 477 | 189 | 7 582 | 662 | 6 014 | 12 221 |
| 135–144 | 2 032 | 1 217 | 926 | 3 653 | 900 | 1 682 | 6 974 | 2 819 | 6 319 | 14 464 |
| 145–154 | 144 | 174 | 30 | 207 | 39 | 182 | 1 127 | 681 | 1 386 | 3 038 |
| 155-164 | 343 | 310 | 49 | 607 | 125 | 117 | 1 378 | 787 | 1 431 | 2 036 |
| 165–174 | 6 | 36 | 9 | 64 | 0 | 0 | 166 | 29 | 158 | 562 |
| 175–184 | 217 | 0 | 0 | 5 | 0 | 7 | 173 | 0 | 31 | 119 |
| 185–194 | 191 | 38 | 0 | 5 | 0 | 0 | 14 | 28 | 28 | 142 |
| 195+ | 0 | 0 | 0 | 3 | 0 | 0 | 0 | 2 | 10 | 46 |

Table 2.2: Softwood volumes ($m^3\,ha^{-1}$) by age and conifer content

| Age(yrs) | Conifer Content(%) | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | 0–10 | 11-20 | 21–30 | 31–40 | 41–50 | 51–60 | 61–70 | 71–80 | 81–90 | 91–100 |
| 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 10 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 20 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 30 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 40 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 50 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 1 | 1 |
| 60 | 0 | 0 | 0 | 0 | 0 | 0 | 13 | 25 | 38 | 51 |
| 70 | 0 | 7 | 14 | 21 | 27 | 34 | 50 | 66 | 82 | 98 |
| 80 | 0 | 15 | 30 | 45 | 60 | 75 | 92 | 109 | 125 | 142 |
| 90 | 0 | 22 | 45 | 67 | 90 | 112 | 130 | 147 | 165 | 182 |
| 100 | 0 | 29 | 58 | 87 | 116 | 145 | 164 | 182 | 201 | 219 |
| 110 | 0 | 35 | 70 | 104 | 139 | 174 | 194 | 213 | 233 | 252 |
| 120 | 0 | 40 | 80 | 1120 | 160 | 200 | 220 | 241 | 261 | 282 |
| 130 | 0 | 45 | 89 | 134 | 178 | 223 | 244 | 265 | 287 | 308 |
| 140 | 0 | 49 | 97 | 146 | 195 | 243 | 265 | 287 | 310 | 332 |
| 150 | 0 | 52 | 104 | 157 | 209 | 261 | 284 | 307 | 330 | 353 |
| 160 | 0 | 55 | 111 | 166 | 222 | 277 | 301 | 324 | 348 | 371 |
| 170 | 0 | 58 | 117 | 175 | 233 | 292 | 316 | 340 | 364 | 388 |
| 180 | 0 | 61 | 122 | 183 | 244 | 305 | 329 | 354 | 379 | 403 |
| 190 | 0 | 63 | 127 | 190 | 254 | 317 | 342 | 368 | 393 | 418 |
| 200 | 0 | 66 | 132 | 198 | 264 | 330 | 356 | 382 | 407 | 433 |

Table 2.3: Hardwood volumes ($m^3 ha^{-1}$) by age and conifer content.

| Age(yrs) | Conifer Content(%) | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | 0–10 | 11-20 | 21–30 | 31–40 | 41–50 | 51–60 | 61–70 | 71–80 | 81–90 | 91–100 |
| 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 10 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 20 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 30 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 40 | 43 | 32 | 21 | 11 | 0 | 0 | 0 | 0 | 0 | 0 |
| 50 | 99 | 80 | 61 | 42 | 23 | 18 | 14 | 9 | 5 | 0 |
| 60 | 144 | 120 | 96 | 72 | 48 | 39 | 29 | 19 | 10 | 0 |
| 70 | 180 | 153 | 127 | 100 | 73 | 59 | 44 | 29 | 15 | 0 |
| 80 | 209 | 181 | 153 | 124 | 96 | 77 | 58 | 38 | 19 | 0 |
| 90 | 233 | 204 | 174 | 145 | 116 | 93 | 69 | 46 | 23 | 0 |
| 100 | 253 | 222 | 192 | 162 | 132 | 105 | 79 | 53 | 26 | 0 |
| 110 | 269 | 238 | 207 | 176 | 145 | 116 | 87 | 58 | 29 | 0 |
| 120 | 283 | 251 | 219 | 187 | 155 | 124 | 93 | 62 | 31 | 0 |
| 130 | 295 | 262 | 229 | 196 | 163 | 130 | 98 | 65 | 33 | 0 |
| 140 | 306 | 271 | 237 | 203 | 168 | 135 | 101 | 67 | 34 | 0 |
| 150 | 315 | 279 | 244 | 208 | 173 | 138 | 104 | 69 | 35 | 0 |
| 160 | 322 | 286 | 249 | 213 | 176 | 141 | 106 | 70 | 35 | 0 |
| 170 | 329 | 292 | 254 | 216 | 178 | 142 | 107 | 71 | 36 | 0 |
| 180 | 336 | 297 | 258 | 219 | 180 | 144 | 108 | 72 | 36 | 0 |
| 190 | 341 | 301 | 261 | 220 | 180 | 144 | 108 | 72 | 36 | 0 |
| 200 | 347 | 305 | 264 | 222 | 180 | 144 | 108 | 72 | 36 | 0 |

Table 2.4: Proportion of disturbed stands regenerating to conifer content classes.

| Conifer Content | Proportion |
|---|---|
| 0–10 | 0.5500 |
| 11–20 | 0.1738 |
| 21–30 | 0.1018 |
| 31–40 | 0.0125 |
| 41–50 | 0.0739 |
| 51–60 | 0.0134 |
| 61–70 | 0.0233 |
| 71–80 | 0.0486 |
| 81–90 | 0.0028 |
| 91-100 | 0.0000 |

The disturbance schedule for sustained yield management (Scenario SY) was developed using a variant of the area–volume check (Davis, 1966) developed by the author. A standard area–volume check takes as input a table of forest area by age class and a yield table. Through an iterative procedure it finds the even-flow level of harvest which will result in the entire forest area being cut over exactly during a specified rotation period. This model rations out "surplus" old-growth volume over the rotation period.

Our variant of the area–volume check allows for the multiple yield tables and succession assumptions required by the growth models used in the NDM representation described below. I also allow for land base discrimination. Stands are classed as part of the hardwood land base if the conifer content is 50 percent or less; otherwise they are part of the softwood land base. In hardwood stands, even flow of hardwood volume is required; softwood volume is treated as incidental. The opposite applies to softwood stands. I chose to analyze a rotation of 100 years. This scenario is intended to represent disturbance levels under the sustained yield management model in Alberta.

The view of NDM management examined here is that the selection of stands for harvest should approximate nature's "selection" of stands for stand-replacing fires. This requires a model of the probability of occurrence of stand-replacing fire in a particular stand under natural conditions (*i.e.* without the effect of fire suppression).

Murphy (1985) applies the methods of Van Wagner (1978) to estimate disturbance rates from inferred age-class distributions of Alberta's forests as of the years 1969, 1949, 1929, and 1909. He uses the disturbance rate as of 1949 to represent the pre-suppression disturbance rate. If one assumes that photo-interpreted age accurately represents years since disturbance and that the probability of a stand-replacing fire is independent of stand characteristics, the age class distribution can be described using a negative exponential probability density function. The parameter of the distribution represents the annual probability of any hectare in the forest being burnt by a stand-replacing fire. Based on the inferred age class distribution as of 1949, Murphy estimates a fire probability for Alberta's northern forest of 0.0196. I use a disturbance rate of 0.02 to approximate this pre-suppression rate in Scenario M2.

Using the inventory as of 1969, Murphy estimates a fire probability of 0.011. In scenario M1, I approximate this with a burn rate of 0.01. This rate is examined because it represents an intermediate level between Murphy's and Cumming's pre-suppression rates of disturbance. The fire return interval implied by the one percent disturbance rate also corresponds with the 100-year return interval explicit in the sustained yield scenario.

Cumming (1997) develops his estimates of fire probability from maps of the areal extent of all known fires in his study area between the years 1940 and 1993. Based on statistical models of the types of stands burned in the years 1980 through 1993, he develops models of the composition of fires and thus stand type specific disturbance rates. He uses a statistical model based on changes in the fire size distribution to correct for the effects of fire suppression. He differentiates between background years and event years, where event years are those years with extreme fire rates; the years 1940–1941 and 1980–1982 are the event years in Cumming's analysis. Fire events are assumed to occur every 30 years or so, and last an average of 2 to 3 years. Disturbance probabilities based on the conifer (specifically white spruce) content of stands are presented in Table 2.5. The average disturbance rates from this table are used in scenario C1.

Table 2.5: Mean disturbance rates based on conifer content.

| | Type of Fire Year | | |
|---|---|---|---|
| Conifer Content | Average | Background | Event |
| <10 | 0.0027 | 0.00174 | 0.0104 |
| 10-20 | 0.0027 | 0.00174 | 0.0121 |
| 20-30 | 0.0027 | 0.0017 | 0.0139 |
| 30-40 | 0.0027 | 0.0017 | 0.0156 |
| 40-50 | 0.003 | 0.0017 | 0.0173 |
| 50-60 | 0.003 | 0.0017 | 0.0211 |
| 60-70 | 0.003 | 0.0017 | 0.0249 |
| 70-80 | 0.0045 | 0.0017 | 0.0287 |
| 80-90 | 0.0055 | 0.00408 | 0.0325 |
| >90 | 0.008 | 0.00408 | 0.0363 |

## 2.3 Simulation Results

The annual area disturbed under each of the four scenarios is presented in Figure 2.1. In scenario M1, 7 850 ha of forest is disturbed each year. In scenario M2, 15 700 ha are disturbed each year. There is no variation in annual area disturbed under these scenarios because they assume constant rates of disturbance independent of forest composition or age-class structure. In scenario SY, the annual area disturbed closely approximates that in scenario M1, although there is some slight variation between years. In scenario C1, the annual area disturbed starts at about 3 500 ha and declines very slightly over the 100 year planning horizon. The rate of disturbance for scenario C1 is less than half that in the other scenarios.

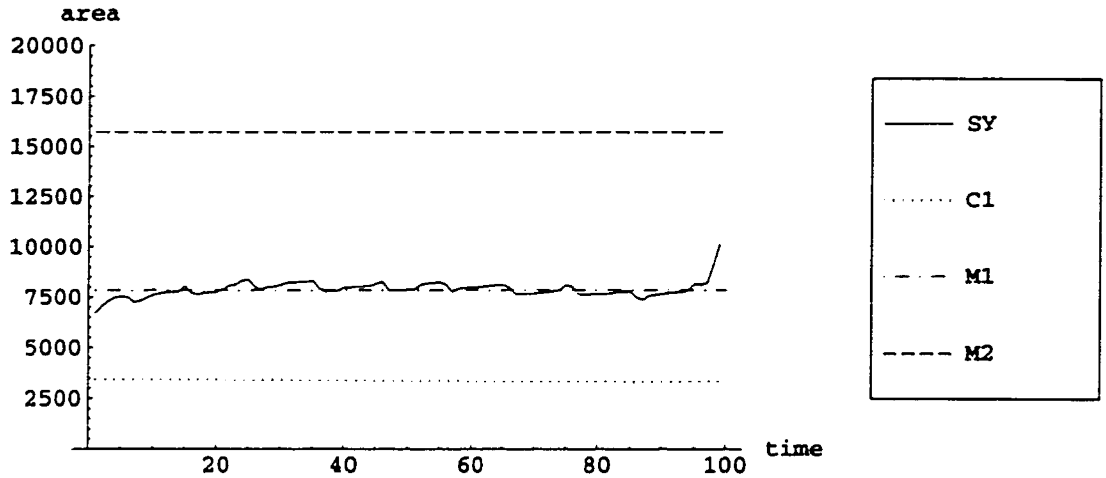


Figure 2.1: Annual area disturbed.

The softwood and hardwood volumes disturbed annually are presented in Figures 2.2 and 2.3 respectively. The sawtooth pattern apparent in these figures is due to the way volumes are assigned to the inventory. Volumes are tabulated by 10-year age classes. In each year of the planning horizon, each inventory record is assigned a volume based on age rounded to the nearest decade, leading to jumps in volume disturbed. Under the SY scenario, approximately $1.6 \times 10^6$ $m^3$ of softwood and $850 \times 10^3$ $m^3$ of hardwood are disturbed each year of the planning horizon. Under the M1 scenario, between $600 \times 10^3$ and $1.0 \times 10^6$ $m^3$ of softwood and $400 \times 10^3$ to $700 \times 10^3$ $m^3$ of hardwood are disturbed annually. In scenario M2, the disturbed softwood volume decreases from $1.2 \times 10^6$ $m^3$ to about $800 \times 10^3$ $m^3$ over the planning horizon. The disturbed hardwood volume decreases from $1.3 \times 10^6$ $m^3$ to a low of about $550 \times 10^3$ $m^3$. In scenario C1, the softwood volume disturbed increases from about $350 \times 10^3$ $m^3$ to about $850 \times 10^3$ $m^3$ over the planning horizon. The hardwood volume disturbed is nearly constant at about $200 \times 10^3$ $m^3$.

Even though the annual areas disturbed under scenarios M1 and SY are similar, the volume disturbed in scenario M1 is about half that in SY. This occurs because in scenario SY area is scheduled for harvest on an oldest (*i.e.* highest volume) first basis, and in scenario M1 areas are disturbed proportionally across the age class distribution.

The average volume per hectare disturbed in each year of the planning horizon is presented in Figure 2.4. For most of the planning horizon, an average of about 310

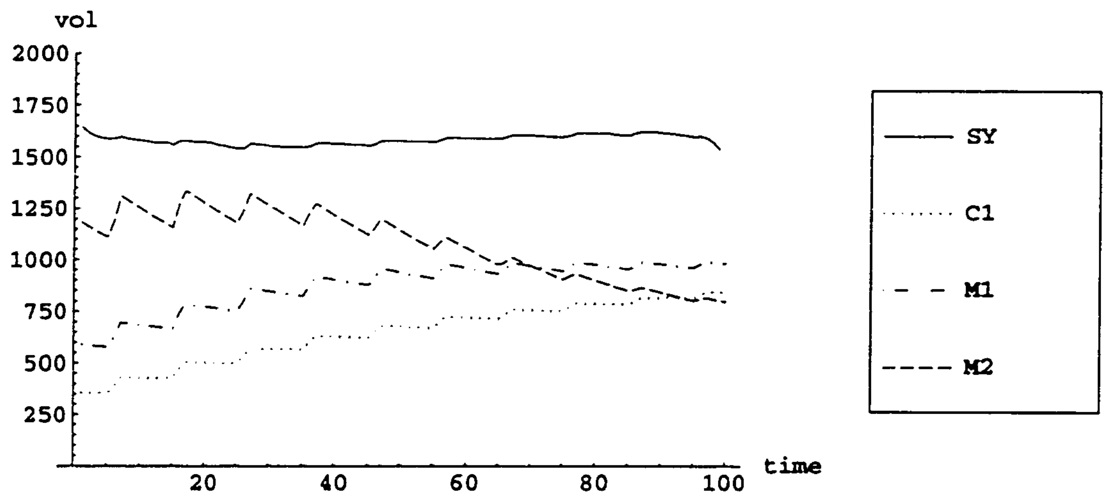


Figure 2.2: Softwood volume disturbed.

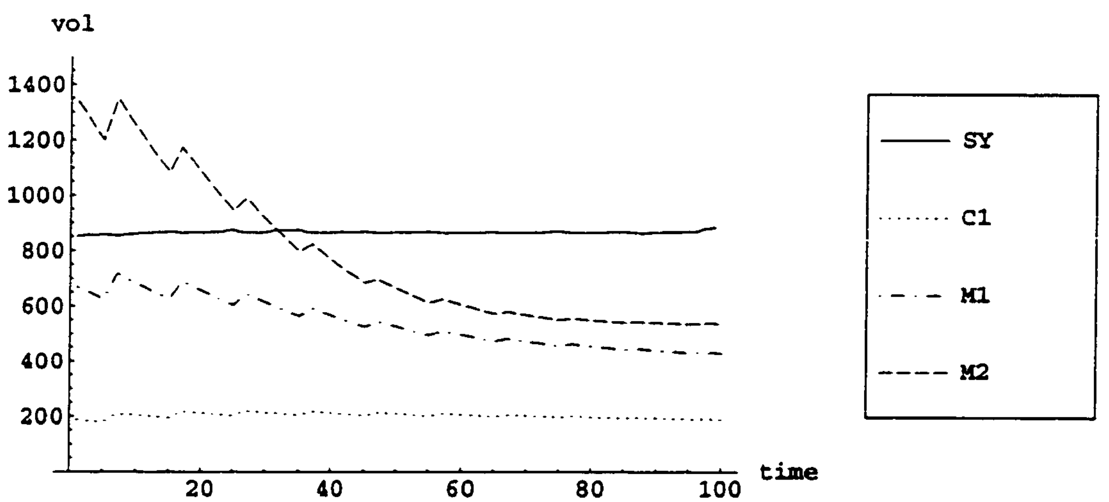


Figure 2.3: Hardwood volume disturbed.

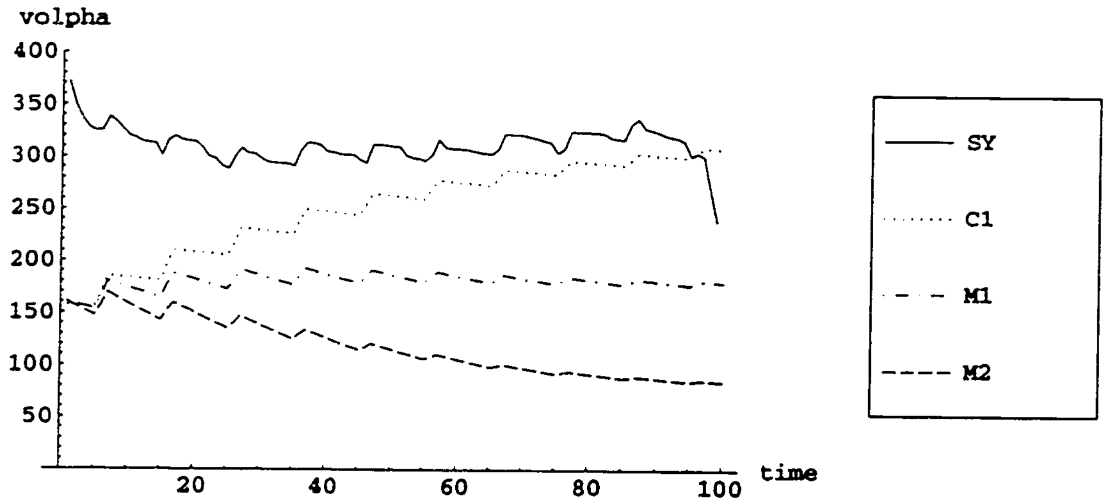


Figure 2.4: Average volume disturbed per hectare.

$m^3\,ha^{-1}$ is harvested under scenario SY. At the beginning of the planning horizon, nearly 370 $m^3\,ha^{-1}$ are disturbed; at the end, about 240 $m^3\,ha^{-1}$ are disturbed. These changes reflect the "rationing out" of overmature volume by the area-volume check procedure over the planning horizon. The 240 $m^3\,ha^{-1}$ disturbed at the end of the planning horizon is very near the long run average for this forest managed on a 100-year rotation. In scenario M1, the average volume disturbed is fairly stable over the planning horizon at about 180 $m^3\,ha^{-1}$. In scenario M2 the average volume disturbed declines from about 160 $m^3\,ha^{-1}$ to 90 $m^3\,ha^{-1}$. In scenario C1 the average volume disturbed increases from about 160 to about 300 $m^3\,ha^{-1}$ over the planning horizon. The increase in disturbed volume in scenario C1 is related to the small annual area disturbed: more of the forest gets a chance to age.

## 2.4 Discussion

The results presented above provide some starting points for a discussion of disturbance-based management. There should not be too much weight placed on the numerical results: the important information is the relative magnitudes. In this section, I raise a number of questions in order to stimulate discussion.

Perhaps the most striking differences in the results presented here are between the two pre-suppression natural disturbance regimes: Murphy's two percent model (scenario M2) and Cumming's conifer content specific model (scenario C1). The annual area disturbed in scenario M2 is 4.5 times as great as that disturbed in scenario C1. Given the differences in estimated disturbance rates, how does a manager decide which rate, if any, is appropriate for management? It could be that one of the rates is "more correct" than the other and that further investigation could help determine which is the appropriate rate. It is also possible that the mean rates reported by Cumming and Murphy are simply different outcomes of the same stochastic process. Chapter 3 and a version published as Armstrong (1999) present some evidence that this is the case In a Monte Carlo simulation of annual area burned, the mean area burnt from a pair of randomly generated 50 year series differed by a multiplicative factor of four, in one out of three draws.

If the rates proposed by Cumming are close to correct, many of the stands in the study area must be much older than is recorded in the forest inventory. It is important to make the distinction between stand age and tree age. Stand age is the number of years between the most recent stand-replacing disturbance and the present time. Individual trees within a stand could have germinated or sprouted in the interim. The age recorded on the inventory records is determined through photo-interpretation, typically by estimating the height and the species of the canopy, assigning a site productivity class to the stand, and using height-age curves indexed by site class to stand age. The resulting estimate is more directly related to the age of the canopy trees than it is to the number of years since a stand-replacing event.

Divergence between photo-interpreted stand age and the number of years since a stand-replacing event could occur for a number of reasons. It is possible that white spruce could germinate in an aspen stand a number of years after the stand-replacing event. Through succession, white spruce could eventually dominate the stand. The photo-interpreted age of such a stand would reflect the age of the spruce trees, not accounting for the lag between stand initiation and spruce germination. Another possibility is that some aspen stands develop into uneven-aged stands through a process of gap phase dynamics (Shugart, 1984). Again the photo-interpreted age would be based on the age of the canopy trees which would be in a continual state of replacement, resulting in an underestimate of stand age. Cumming (1997) presents some evidence that this occurs in the study area.

Under Cumming's natural disturbance regime, the annual area disturbed would be 22 percent of that suggested by Murphy's 2% natural disturbance regime, and about 45 percent of that scheduled under sustained yield management with a 100 year rotation. This clearly has some important timber supply implications, especially if the forest is in transition from sustained yield management to natural disturbance based management. In Alberta, most of the boreal forest has been allocated to industry through long term forest management agreements: there is little or no room to increase harvest levels in any part of the province given natural timber growth rates. Large scale industrial development has occurred on the assumption of a timber supply level appropriate to sustained yield management. Would society be willing to pay for the perceived benefits of NDM management with a drop in harvestable area and economic activity of such magnitude? Alternatively, would society be willing to accept intensive management on part of the land base to make up the shortfall (if possible)?

The effects of strict NDM management are even more pronounced when we consider annual harvest volume rather that annual harvest area. This occurs because in the sustained yield scenario, the oldest (and by our yield tables, highest volume) stands are disturbed first. In the natural disturbance models examined here, the disturbance rate is independent of age[2], so age classes are being disturbed in proportion to their representation in the inventory. This means that very young stands with no merchantable volume will be scheduled for disturbance. The effects of this show up in the lower average volumes per hectare disturbed (Figure 2.4) under the natural disturbance scenarios.

This raises another important point. Nature would "schedule" some low volume, low value stands for disturbance. How important is it to emulate these "unprofitable" disturbances in a natural disturbance model? At the extreme, it may be desirable from an ecological point of view to deliberately burn or otherwise disturb some areas of immature forest in order to simulate the natural disturbance regime. This deliberate disturbance would be required to offset the reduction in fire rates due to fire suppression. Who should pay for the additional cost of these objectives? Timber harvest occurs only on the net productive land base. If the rate of disturbance on the non-productive area has been reduced through fire suppression efforts, should there be planned disturbances (e.g. prescribed burning) introduced to the unproductive land base?

Another characteristic of the natural fire regime is the variability in the disturbance rate from year to year. Given this variability, and assuming that the forest products industry requires some stability in the year to year flow of timber, which rate is the appropriate one on which to base management: the overall mean, the mean for background years, the mean for event years, or some other statistic (e.g. the median)? This becomes a risk–risk trade off decision. If one chooses too high a rate for timber harvest under a natural disturbance model, then some ecological benefits may not occur. If too low a rate is chosen, perhaps the risk of fire and severity of fires will increase, also reducing ecological benefits. Perhaps "the rate" is not an appropriate objective in any case: it is conceivable that the variability in annual area disturbed is as important ecologically as the average.

Given the difficulty in determining the average natural rate of disturbance, and the extreme year to year variability in the rate, the natural disturbance rate should not be the sole determinant of harvest rate. Other alternatives include variable rotation management strategies (Stelfox, 1995), or using models which constrain the forest structure (defined for example by proportion of area within age class and spruce content) to fall within acceptable bounds (see *e.g.* Cumming *et al.* (1994)). "The rate" or some measure of risk/hazard should be an integral element in forest structure approaches and in determining the expected value of choices: it should not be the only objective of management.

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[2]In Cumming's model, the rate of disturbance changes with conifer content, which changes with age.

As with any modeling exercise, the analysis presented here is a much simplified representation of reality. The spatial elements of disturbance are ignored completely in this study, and they are certainly worth examining. It is conceivable that timber harvest can mimic some forms of natural disturbance other than wildfire (*e.g.* windthrow, insect attack, gap dynamics), but these were not examined here. I have ignored management activities which may affect the probability of disturbance or growth rates (*e.g.* silviculture, changes in fire suppression strategy). Despite these omissions, I believe that I can say that moving from sustained yield to natural disturbance models of forest management can have tremendous implications for the timber supply potential of an area, and that choosing "the natural disturbance rate" appropriate for a forest is both difficult and risky.

# Bibliography


Alberta Forest Service. 1985. Alberta Phase 3 forest inventory: Yield tables for unmanaged stands. ENR Report Dept. 60a, Department of Energy and Natural Resources, Edmonton, Alberta.

Alberta Pacific Forest Industries Inc. undated. Environmental management. [web page]. <http://www.alpac.ca/regional.html> [Accessed 11 June 1998].

Armstrong G.W. 1999. A stochastic characterisation of the natural disturbance regime of the boreal mixedwood forest with implications for sustainable forest management. Can. J. For. Res. **29**: 424–433.

Cumming S.G. 1997. Landscape dynamics of the boreal mixedwood forest. Ph.D. dissertation, Department of Forest Sciences. University of British Columbia, Vancouver.

Cumming S.G., Burton P., Prahacs S., and Garland M. 1994. Potential conflicts between timber supply and habitat protection in the boreal mixedwood of Alberta, Canada: A simulation study. Forest Ecology and Management **68**: 281–302.

Cumming S.G., Burton P.J., Joy M., Klinkenberg B., Schmiegelow F.K.A., and Smith J.N.M. 1995. Experimental habitat fragmentation and simulation of landscape dynamics in the boreal mixedwood: A pilot study. Canada-Alberta Partnership Agreement in Forestry Report A5016-137, Canadian Forest Service and the Alberta Land and Forest Service.

Davis K.P. 1966. Forest Management: Regulation and Valuation. McGraw-Hill Book Company, New York, 2nd edn.

Joy M.W. 1995. Handling uncertainty in GIS and environmental models: an application in forest management. Master's thesis, Department of Geography, University of British Columbia, Vancouver, B. C.

Murphy P.J. 1985. Methods for evaluating the effects of forest fire management in Alberta. Ph.D. dissertation, Department of Forestry. University of British Columbia, Vancouver.

Reed W.J. and Errico D. 1986. Optimal harvest scheduling at the forest level in the presence of risk of fire. Can. J. For. Res. **16**: 266–278.

Shugart H.H. 1984. A Theory of Forest Dynamics: The Ecological Implications of Forest Succession Models. Springer, New York.

Stelfox J.B. (ed.). 1995. Relationships between stand age, stand structure, and biodiversity in aspen mixedwood forests in Alberta, Edmonton, Alberta. Jointly published by Alberta Environmental Centre (AECV95-R1), Vegreville Alberta and Canadian Forest Service (Project No. 0001a).

Van Wagner C.E. 1978. Age-class distribution and the forest fire cycle. Can. J. For. Res. **8**: 220–227.

Van Wagner C.E. 1983. Simulating the effect of forest fire on long-term annual timber supply. Can. J. For. Res. **13**: 451–457.

# Chapter 3

# A Stochastic Characterization of the Natural Disturbance Regime[1]

## 3.1 Introduction

The determination of annual harvest area is possibly the most important forest level decision a forest manager can make. This decision determines how much timber the forest supplies annually and will, at least partially, determine the future age class structure of the forest: the decision has major financial and ecological implications. This determination is, of course, a management decision: social, economic, and ecological factors may be considered. Under NDM management, one could argue that the sum of the annual area harvested and burned in each age class of a managed forest should equate to the area that would have burned under natural conditions. Therefore, in order to decide annual harvest area under NDM management, the forest manager must be aware of the annual burn rate given current fire prevention and suppression efforts, and the annual burn rate that would have applied under natural conditions. Alberta-Pacific Forest Industries Inc. is planning to use this model to help determine harvest rates for their Forest Management Agreement area (FMA) in northeastern Alberta. They plan research to "determine the differences in disturbance rates across the FMA, so that harvest rates and methods can emulate the disturbance rate for each 'disturbance rate' zone" (Alberta Pacific Forest Industries Inc., undated). For this company, an estimate of the rate of natural disturbance is information critical to their forest management strategy.

Johnson and Gutsell (1994) define some terminology useful for describing rates of disturbance. The fire cycle is the time required to burn an area equal in size to the area of interest. A numerically equivalent concept is the average fire interval, or the expected time between fires at any point in the forest. Assuming that the probability of fire is independent of stand age, the annual proportion of area burned or the burn rate is the

[1] A version of this chapter has been published. Armstrong, G.W. 1999. A stochastic characterisation of the natural disturbance regime of the boreal mixedwood forest with implications for sustainable forest management. Can. J. For. Res. 29:424–433.

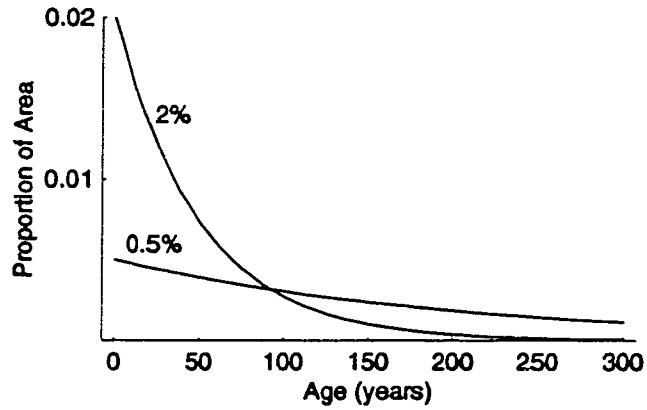


Figure 3.1: Stand age distributions resulting from constant burn rates of 2% and 0.5%.

inverse of the fire cycle.

Existing estimates of annual natural burn rates are highly variable. Turner and Romme (1994) summarize the fire cycles reported in number of different studies of forests in North America and Europe. For areas that are part of the boreal forest, they report fire cycles ranging from 26 to 500 years, corresponding with annual burn rates of 3.8% and 0.2%. These differences are attributed to continental, latitudinal, and elevational gradients. There are other estimates of annual burn rates specific to the boreal mixedwood forest of Canada. Kabzems *et al.* (1976) estimated the annual burn rate in the boreal mixedwood section of Saskatchewan to be 0.32%. Murphy (1985), using the technique developed by Van Wagner (1978), estimated the annual natural rate of burn in the predominantly mixedwood forest of northern Alberta to be 2%. Cumming (1997), using 54 years of fire history data, estimated the annual natural burn rate on a large subset ($7.4 \times 10^6$ ha) of Murphy's study area to be about 0.5%.

The selection of the disturbance rate for NDM management will clearly have important implications for harvest levels and the age-class structure of the resulting forest. Figure 3.1 presents the equilibrium negative exponential age class distribution (see Van Wagner, 1978) that would result from the disturbance rates estimated by Murphy (1985) and Cumming (1997). The Van Wagner (1978) age class distribution model holds when the annual burn rate is constant and the spatial location of fires is random. The two rates were developed for similar forests, yet differ by a factor of four. Which of these rates best describes the natural disturbance regime of the forest?

An alternative approach to setting harvest rates under NDM management would be to specify the characteristics of the desired future forest, and to define a management schedule which would tend to drive the forest towards that structure. Weyerhauser Canada Ltd. has developed such a strategy for its Forest Management License Agreement area (FMLA) in northern Saskatchewan. Their goal is the negative exponential forest age distribution that would arise from a 70-year fire cycle (Weyerhaeuser Canada, 1997).

If either of these methods is used to set harvest rates, the decision maker is implicitly assuming that an estimate of the appropriate disturbance rate or age class structure for the forest can be developed. It is unclear whether this is possible when large variability in annual area burned is characteristic of the natural disturbance regime. This study has three main objectives: to develop a characterisation of the natural fire regime that incorporates annual variability in area burned, to examine how this characterisation affects the precision of estimates of average annual burn rates, and to examine the age class distributions that result from the application of this fire regime.

The techniques used in this study consist of a simple statistical analysis to characterize the natural fire regime, and Monte Carlo simulations to examine the effects of the assumed fire regime on the precision of estimates of average burn rate and on forest age class distributions.

## 3.2 Characterisation of Annual Burn Rates

A roughly rectangular study area in Alberta (bounded approximately by 55°1′ N and 57°42′ N latitude, and 110° W and 115° W longitude) was chosen as the study area. This area is nearly coincident with the study area used by Cumming (1997) and has a total area of about $8.6 \times 10^6$ ha. The location of the study area is shown in Figure 3.2. Most of the study area is contained in the Mixedwood section (B.18a) of Rowe's Boreal forest region (Rowe, 1972). The study area supplies a large pulp mill with both hardwood and softwood pulp logs, and several sawmills with softwood saw logs. The pulp mill started operation in 1993. Most of the sawmills have been operating for much longer. The tree species of commercial importance are trembling aspen (*Populus tremuloides* Michx.), balsam poplar (*Populus balsamifera* L.), white spruce (*Picea glauca* (Moench) Voss), jack pine (*Pinus banksiana* Lamb.), and black spruce (*Picea mariana* (Mill.) B.S.P.).

The Government of Alberta's Department of Environmental Protection maintains an online database of all forest fires detected in the province since 1961 (Government of Alberta, 1997). The database includes the location of fire ignition, fire cause, fire size, and other useful information. From this database, the total annual area of all lightning caused fires that originated in the study area from 1961 through 1995 was calculated. Only lightning caused fires were considered in order to limit the study to naturally caused fires. It could be argued that the natural fire regime should incorporate the fires that would have been set (accidentally or for habitat management purposes) by the aboriginal people inhabiting the area before the arrival of the Europeans, but this is not considered here. The total area calculated includes area burned outside the study area by fires originating inside. It is assumed that this is balanced by area burned inside the study area by fires originating outside.

Table 3.1 presents the time series of annual area burned for the study area, as well

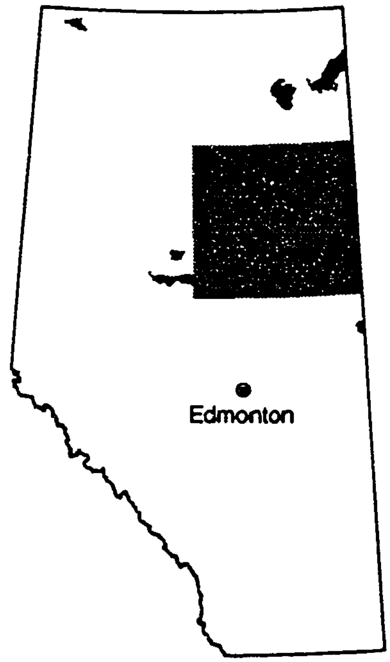


Figure 3.2: Location of study area in Alberta, Canada

as a transformation that controls for fire suppression. The data are also presented as their natural logarithms.

Cumming (1997) presents evidence that fire suppression effort and effectiveness has changed over the time period covered by the data set. He develops correction factors to provide estimates of the area that would have burned if no fire suppression took place. The factors were developed on the basis of a statistical analysis of the size distributions of lightning caused fires for the period 1961–1993. He identifies four periods (1961–1970, 1971–1979, 1980–1982, and 1983–1993) with different levels of fire suppression effectiveness. In general, fire suppression effectiveness has been improving over time. Cumming attributed the temporary reduction in effectiveness for the period 1980–1982 to the extreme nature of the fire events in that period: the fire suppression resources were overwhelmed by the number of fires in that period. I assume that 1994 is similar to the period 1983–1993, and that 1995 is similar to the period 1980–1982. The area that would have burned under pre-suppression conditions is estimated by multiplying the area actually burned by the correction factor. A primary goal of this paper is the characterisation of the natural fire regime, so much of the analysis focuses on the pre-suppression time series.

The untransformed time series of annual area burned is presented in Figure 3.3. The time series is dominated by large areas burned between 1980 and 1982 and again in 1995. Cumming (1997) categorizes the period 1980–1982 as a fire event due to the

Table 3.1: Annual area burned and its natural logarithm. The columns labeled "Uncorrected" refer to the observed area burned. "Pre-Suppression" areas are corrected for the effect of fire suppression using the correction factor. The estimated means and standard deviations are recorded in the rows $\hat{\mu}$ and $\hat{\sigma}$.

| | Correction | Area (ha) | | Natural Logarithm | |
|---|---|---|---|---|---|
| Year | Factor | Uncorrected | Pre-Suppression | Uncorrected | Pre-Suppression |
| 1961 | 1.00 | 11 470.39 | 11 470.39 | 9.348 | 9.348 |
| 1962 | 1.00 | 7.69 | 7.69 | 2.040 | 2.040 |
| 1963 | 1.00 | 1 155.49 | 1 155.49 | 7.052 | 7.052 |
| 1964 | 1.00 | 1 767.58 | 1 767.58 | 7.477 | 7.477 |
| 1965 | 1.00 | 369.96 | 369.96 | 5.913 | 5.913 |
| 1966 | 1.00 | 3 573.66 | 3 573.66 | 8.181 | 8.181 |
| 1967 | 1.00 | 28.33 | 28.33 | 3.344 | 3.344 |
| 1968 | 1.00 | 4.45 | 4.45 | 1.493 | 1.493 |
| 1969 | 1.00 | 3 094.67 | 3 094.67 | 8.037 | 8.037 |
| 1970 | 1.00 | 5 513.72 | 5 513.72 | 8.615 | 8.615 |
| 1971 | 2.61 | 9 891.17 | 25 815.96 | 9.199 | 10.159 |
| 1972 | 2.61 | 6 519.66 | 17 016.31 | 8.783 | 9.742 |
| 1973 | 2.61 | 25.09 | 65.49 | 3.222 | 4.182 |
| 1974 | 2.61 | 2 578.86 | 6 730.81 | 7.855 | 8.814 |
| 1975 | 2.61 | 69.20 | 180.62 | 4.237 | 5.196 |
| 1976 | 2.61 | 147.71 | 385.52 | 4.995 | 5.955 |
| 1977 | 2.61 | 38.04 | 99.29 | 3.639 | 4.598 |
| 1978 | 2.61 | 3 075.41 | 8 026.81 | 8.031 | 8.991 |
| 1979 | 2.61 | 736.27 | 1 921.68 | 6.602 | 7.561 |
| 1980 | 1.85 | 62 150.24 | 114 977.94 | 11.037 | 11.652 |
| 1981 | 1.85 | 382 795.75 | 708 172.15 | 12.855 | 13.470 |
| 1982 | 1.85 | 183 338.14 | 339 175.56 | 12.119 | 12.734 |
| 1983 | 4.84 | 42.50 | 205.70 | 3.750 | 5.326 |
| 1984 | 4.84 | 9 605.60 | 46 491.10 | 9.170 | 10.747 |
| 1985 | 4.84 | 259.60 | 1 256.46 | 5.559 | 7.136 |
| 1986 | 4.84 | 767.60 | 3 715.18 | 6.643 | 8.220 |
| 1987 | 4.84 | 171.20 | 828.61 | 5.143 | 6.720 |
| 1988 | 4.84 | 697.10 | 3 373.96 | 6.547 | 8.124 |
| 1989 | 4.84 | 83.00 | 401.72 | 4.419 | 5.996 |
| 1990 | 4.84 | 13 393.50 | 64 824.54 | 9.503 | 11.079 |
| 1991 | 4.84 | 1 547.60 | 7 490.38 | 7.344 | 8.921 |
| 1992 | 4.84 | 288.10 | 1394.40 | 5.663 | 7.240 |
| 1993 | 4.84 | 2 428.60 | 11 754.42 | 7.795 | 9.372 |
| 1994 | 4.84 | 2 789.50 | 13 501.18 | 7.934 | 9.511 |
| 1995 | 1.85 | 148 149.46 | 274 076.50 | 11.906 | 12.521 |
| $\hat{\mu}$ | | 24 530.71 | 47 967.66 | 7.013 | 7.871 |
| $\hat{\sigma}$ | | 73 869.89 | 136 379.95 | 2.796 | 2.853 |

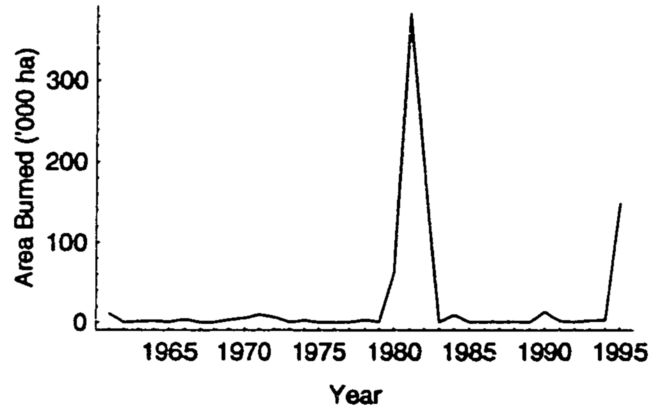


Figure 3.3: Total annual area ($10^3$ ha) of lightning caused fires originating in the study area.

extreme nature of the fires in those years. The annual burn rates for the study area are obviously highly variable. Any landscape level simulation model of the natural disturbance regime for this forest must incorporate assumptions about this variability. One could use one rate describing the arithmetic average of the annual burn rates as in the fire cycle game described by Van Wagner (1978). Boychuk and Perera (1997) describe a scenario involving two average rates (one for low burn rate years and one for high burn rate years) for use in their FLAP-X model. The low or high burn rate is applied to a year based on the outcome of a random number draw. Cumming (1997) also uses two average rates (for background years and event years). A year is determined to be a background year or an event year on the basis of a random draw from a Poisson distribution, and the appropriate rate is applied. None of these approaches is entirely satisfactory. The Van Wagner (1978) approach ignores the large variability apparent in the annual area burned, and the approaches of Boychuk and Perera (1997) and Cumming (1997) rely on arbitrary distinctions between background (low rate) and event (high rate) years. The remainder of this section presents a simple and more complete characterisation of the distribution of annual area burned.

Figure 3.4 presents the same information as Figure 3.3 with the area burned plotted on a logarithmic scale. The differences between event and background years appear far less extreme when viewed this way. From Table 3.1, the estimated mean of the natural logarithm of the untransformed area burned ($\hat{\mu}$) is 7.013 and the standard deviation ($\hat{\sigma}$) is 2.796. Figure 3.5 compares the empirical distribution function (EDF) for the time series with the cumulative distribution function for the lognormal distribution defined by $\hat{\mu}$ and $\hat{\sigma}$. Another way of viewing the correspondence between an observed and a hypothesized distribution is a Quantile-Quantile (Q-Q) plot. Figure 3.6 presents the quantiles for both the uncorrected and pre-suppression annual area burned series plotted against the quantiles for a lognormally distributed variable. A perfect correspondence is indicated by

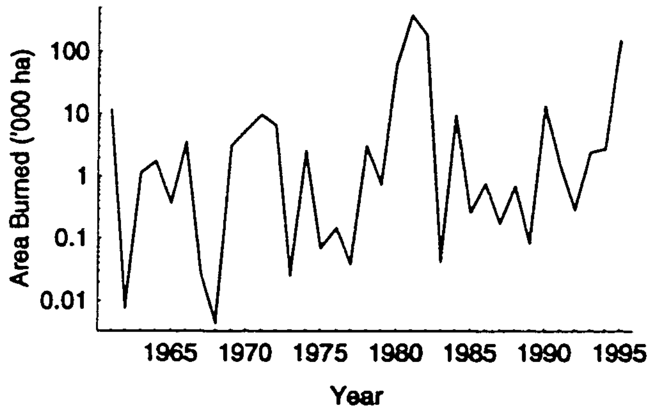


Figure 3.4: Total annual area ($10^3$ ha) of lightning caused fires originating in the study area (logarithmic scale).

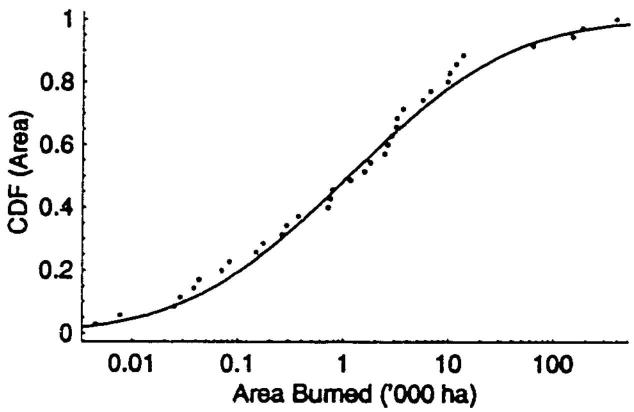


Figure 3.5: Cumulative empirical and lognormal probability density functions for annual area burned.

the 45° line. The fit of the lognormal distribution to the observed data appears good.

If the natural logarithm of a variable is distributed normally, the untransformed variable is distributed lognormally. The probability density function (PDF) of the lognormal distribution is

$$f(x) = (\sqrt{2\pi}\sigma x)^{-1} \exp\left[-\frac{(\ln x - \mu)^2}{2\sigma^2}\right] \tag{3.1}$$

where $x$ refers to the variable of interest (annual area burned in this paper), $\mu$ is the population mean of the natural logarithm of $x$, $\sigma$ is the population standard deviation, and $\sigma^2$ is the population variance (see *e.g.* Ingersoll, 1987). The cumulative probability density function (CDF) of the lognormal distribution is

$$F(x) = \frac{1}{2}\left(1 + \operatorname{erf}\left[\frac{\ln x - \mu}{\sqrt{2}\sigma}\right]\right) \tag{3.2}$$

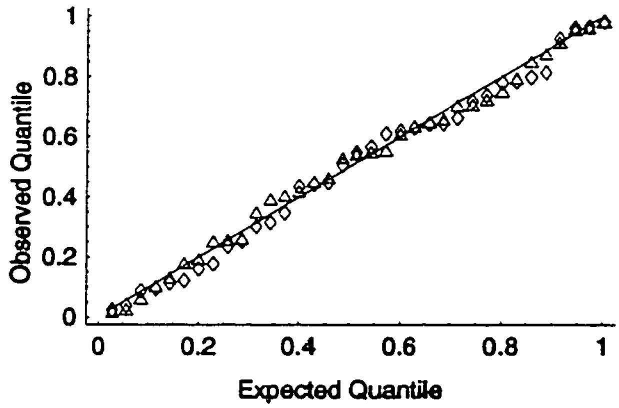


Figure 3.6: Quantile-Quantile plot for annual area burned assuming an expected lognormal distribution. The 45° line represents a perfect fit. The uncorrected series plotted as triangles. The pre-suppression series is plotted as diamonds.

where erf[·] represents the error function. The mean of a lognormal distribution is

$$\bar{x} = \exp\left(\mu + \frac{\sigma^2}{2}\right) \tag{3.3}$$

and the median of the population is calculated as $\exp \mu$.

Two non-parametric goodness-of-fit tests were conducted to test the null hypothesis that the natural logarithm of annual area burned was distributed normally.

The Kolmogorov-Smirnov (K-S) test as described by Sokal and Rohlf (1981) was used to test goodness-of-fit of the normal distribution to the logarithmically transformed burn rates. For distributions expressed as relative frequencies, the critical value for $\alpha = 0.05$ is asymptotically approximated as $D_{.05} = \frac{0.886}{\sqrt{n}}$ when the sample size ($n$) is greater than 30. The null hypothesis is rejected when the greatest absolute difference between the observed and expected cumulative frequencies is greater than the critical value. The maximum absolute difference between the EDF for the natural logarithm of the uncorrected area burned and the CDF for the normal distribution was 0.0723. The maximum absolute difference for the pre-suppression time series was 0.0729. The critical value for $\alpha = 0.05$ is 0.150 for a sample size of 35, therefore the null hypothesis that the natural logarithm of annual area burned is normally distributed is not rejected for either data series.

The Anderson-Darling test as described by Stephens (1986) and implemented by Davis and Stephens (1989) was also used. This goodness-of-fit statistic is better than the K-S test at finding deviations from hypothesized distributions at the tails of the distribution. Because the extreme fire years are likely to have large influences on age class distributions and other characteristics of the forest, it is important that any statistical characterisation fits the tails of the distribution well. The Anderson-Darling test statistic

Table 3.2: Trend detection regression for uncorrected annual area burned time series.

| Coefficient | Estimate | Standard Error | t Statistic | P-value |
|---|---|---|---|---|
| constant | -112.649 | 91.618 | -1.230 | 0.228 |
| year | 0.060 | 0.046 | 1.306 | 0.201 |

Table 3.3: Trend detection regression for pre-suppression annual area burned time series.

| Coefficient | Estimate | Standard Error | t Statistic | P-value |
|---|---|---|---|---|
| constant | -216.388 | 87.570 | -2.471 | 0.019 |
| year | 0.113 | 0.044 | 2.561 | 0.015 |

is

$$A^2 = -n - \frac{1}{n}\sum_{i=1}^{n}(2i-1)\left(\ln F(x_i) + \ln(1 - F(x_{n-i+1}))\right) \tag{3.4}$$

where $F(x_i)$ is the CDF for the hypothesized probability distribution evaluated at the $i$th observation of the data set ($x_i$). When the hypothesized distribution is the normal distribution and both the mean and the variance are estimated from the data, the test statistic is modified to

$$A^* = A^2(1 + \frac{0.75}{n} + \frac{2.25}{n^2}). \tag{3.5}$$

This test statistic requires that the data are sorted by $x_i$. For the logarithmically transformed uncorrected data, the calculated value for $A^*$ is 0.21 and the probability level is 0.14. For the logarithmically transformed pre-suppression area burned, $A^*$ is 0.16 and the probability level is 0.95. The null hypothesis of normality is not rejected for either data set.

Ordinary least squares (OLS) regression was used to test for a trend in annual area burned. The natural logarithm of annual area burned (both the uncorrected and pre-suppression data) was regressed against the year of observation. The trend detection results for the uncorrected and pre-suppression data series are summarized in Tables 3.2 and 3.3 respectively. No trend was detected in the uncorrected time series, but the year coefficient in the pre-suppression time series is significant at the 95% level indicating there may be an upward trend in annual area burned, after adjusting for the effects of fire suppression.

It is possible that the area burned in any given year is related to the area burned in previous years. Perhaps extreme fire years tend to be clumped together such as was observed in 1980–1982. Alternatively, perhaps low fire years tend to follow high fire years due to reduction in available fuel. These possibilities were tested for using OLS regression

Table 3.4: Lagged regression results for uncorrected area burned.

| Coefficient | Estimate | Standard Error | t Statistic | P-value |
|---|---|---|---|---|
| $\hat{\beta}_0$ | 6.322 | 2.056 | 3.074 | 0.005 |
| $\hat{\beta}_1$ | 0.191 | 0.196 | 0.975 | 0.338 |
| $\hat{\beta}_2$ | 0.085 | 0.186 | 0.457 | 0.651 |
| $\hat{\beta}_3$ | -0.166 | 0.183 | -0.907 | 0.372 |

Table 3.5: Lagged regression results for pre-suppression area burned.

| Coefficient | Estimate | Standard Error | t Statistic | P-value |
|---|---|---|---|---|
| $\hat{\beta}_0$ | 6.061 | 1.607 | 3.771 | 0.001 |
| $\hat{\beta}_1$ | 0.051 | 0.179 | 0.286 | 0.777 |
| $\hat{\beta}_2$ | 0.181 | 0.166 | 1.091 | 0.285 |
| $\hat{\beta}_3$ | -0.015 | 0.164 | -0.092 | 0.927 |

to estimate the coefficients for the third order autoregressive process

$$\ln A_t = \beta_0 + \sum_{i=1}^{3} \beta_i \ln A_{t-i} + \epsilon_t \tag{3.6}$$

where $A_t$ represents the area burned in year $t$, the $\beta$s are coefficients, and $\epsilon_t$ is the residual. Tables 3.4 and 3.5 present the results of the OLS estimation for uncorrected and pre-suppression annual area burned. No evidence of serial autocorrelation was detected as none of the coefficients on the lagged variables was significant. The estimated constant term ($\hat{\beta}_0$) was significant in both cases. This simply indicates that the means of the logarithmically transformed data are significantly different from zero.

From the regression analysis, there is no apparent autoregressive relationship in either the uncorrected or pre-suppression annual area burned time series. It is possible that the annual area burned is a result of a purely random or white noise process. The Ljung-Box-Pierce portmanteau test statistic provides a test for a white noise process (Harvey, 1981; White, 1997). The test statistic is

$$Q^* = T(T+2) \sum_{\tau=1}^{P} (T-\tau)^{-1} r_\tau^2 \tag{3.7}$$

where $T$ represent the number of observations in the time series, and $r_\tau$ represents the $\tau$th autocorrelation. For a white noise process, $Q^*$ has an asymptotic $\chi^2_P$ distribution. The autocorrelations are calculated as

$$r_\tau = c_\tau / c_0 \quad \tau = 1, 2, \ldots, P \tag{3.8}$$

Table 3.6: $Q^*$ test statistics for white noise.

| | Uncorrected $lnA_t$ | | Pre-suppression $lnA_t$ | |
|---|---|---|---|---|
| $P$ | $Q^*$ | P-value | $Q^*$ | P-value |
| 1 | 0.43 | 0.511 | 1.42 | 0.233 |
| 2 | 0.71 | 0.701 | 2.49 | 0.288 |
| 3 | 1.41 | 0.704 | 2.50 | 0.475 |
| 4 | 2.29 | 0.683 | 2.63 | 0.621 |
| 5 | 3.25 | 0.662 | 2.72 | 0.743 |
| 6 | 3.27 | 0.774 | 2.97 | 0.812 |
| 7 | 7.13 | 0.415 | 4.41 | 0.732 |
| 8 | 7.13 | 0.523 | 4.56 | 0.803 |
| 9 | 7.30 | 0.606 | 5.03 | 0.831 |
| 10 | 9.53 | 0.483 | 7.68 | 0.660 |
| 11 | 10.50 | 0.486 | 8.72 | 0.648 |
| 12 | 10.56 | 0.567 | 8.72 | 0.727 |

where

$$c_\tau = \frac{1}{T} \sum_{t=\tau+1}^{T} [(x_t - \bar{x})(x_{t-\tau} - \bar{x})] \quad \tau = 0, 1, 2, \ldots, P, \tag{3.9}$$

$x_t$ represents the natural logarithm of area burned in year $t$, and $P$ represents the number of lags one is examining for evidence of residual autocorrelation.

This test statistic was calculated for both the uncorrected and pre-suppression time series of annual area burned using the ARIMA command of the SHAZAM statistical package (White, 1997). This statistic was calculated for $P = 1, 2, \ldots, 12$. The results are presented in Table 3.6. Any lag for which the P-value of $Q^*$ is less than 0.05 would indicate a departure from a white noise process. No such evidence was found.

Because the null hypotheses of normality and lack of serial autocorrelation were not rejected, the annual area burned (as observed and as corrected for fire suppression) in the study area is characterized as a serially independent random draw from a lognormally distributed population. This characterisation is extremely convenient as the lognormal distribution is completely defined by two parameters: the mean ($\mu$) and standard deviation ($\sigma$) of the natural logarithm of annual area burned.

The remaining sections of this paper use this stochastic characterisation of pre-suppression annual area burned in Monte Carlo simulations to examine the variability of estimated burn rates and forest age class distributions. In the simulations, the burned area is represented as the proportion of the study area burned. The mean parameter for the rate-based characterisation can be calculated simply by subtracting the natural logarithm of the total area from the mean parameter for the area-based characterisation ($\mu = 7.871 - \ln(8.6 \times 10^6) = -8.096$). A convenient characteristic of the lognormal distribution is that the standard deviation parameter does not change as a result of this transformation. The Monte Carlo simulations in the following sections draw from a lognormal distribution with $\mu = -8.096$ and $\sigma = 2.853$. Using Equation 3.3 the mean disturbance rate from this distribution is calculated to be 1.78%.

## 3.3 Burn Rate Comparisons

If one chooses to set harvest rate so as to emulate the rate of natural disturbance, the ability to measure the natural disturbance rate is critically important. We have seen in Figure 3.1 that application of the 2% rate estimated by Murphy (1985) and the 0.5% rate estimated by Cumming (1997) would result in very different forests. These rates differ by a factor of four, yet it is unclear which of the two is the more accurate representation of the natural disturbance regime. This problem is examined using Monte Carlo simulation and is approached from two different directions: confidence intervals for the mean rate calculated for different sample periods are developed, as are estimates of the probability that two calculated means can differ by a specified multiplicative factor.

Choice of the sample period from which to estimate mean annual burn rates will depend on the available data and the purpose for which the mean is being estimated. Time intervals used for harvest planning may be relevant. In Alberta, harvest levels are constrained to be within certain limits over a five-year cut control period. The age of available and reliable historical records may limit the sample period. The data used by Cumming (1997) cover a roughly fifty-year time period. Following Heinselman (1973), the forest itself may provide a readable record of disturbance frequency. The oldest interpreted stand age in the study area is 230 years. All three of these sample period lengths are simulated, as well as a 1000 year sample period used to set an upper bound.

A straightforward simulation procedure was used. Let $T$ represent the length of the sample period being simulated. The simulations will be repeated for $T = 5, 50, 230, \text{and } 1000$ years. Let $D$ represent the number of replications ($D = 10\,000$). Create $D \times T$ matrix $B$ populated with annual burn rates randomly drawn from a lognormally distribution population defined by a mean ($\mu$) and variance ($\sigma^2$). The simulations presented here use $\mu = -8.096$ and $\sigma = 2.853$ as determined in the previous section. The burn rate is constrained to be 1 or less to prevent the situation where more than 100% of the study area is burned.

$$b_{dt} = \min(1, \exp x_{dt}), \quad x_{dt} \sim N(\mu, \sigma^2), \quad d = 1, 2, \ldots, D, \quad t = 1, 2, \ldots, T. \tag{3.10}$$

The $D \times 1$ vector $\bar{\mathbf{b}}$ contains the mean burn rate calculated for each replication. It is calculated as

$$\bar{\mathbf{b}} = \frac{1}{T} B\mathbf{i} \tag{3.11}$$

where $\mathbf{i}$ is $T \times 1$ vector of ones. Most of the simulation results are presented as simple descriptive statistics of the data contained in $\bar{\mathbf{b}}$. A statistical summary of the simulation

Table 3.7: Statistical summary of mean burn rate estimates derived from Monte Carlo simulations. The confidence limits and the Pr(2×) and Pr(4×) statistics are explained in the text.

| | Sample Period Length (years) | | | |
|---|---|---|---|---|
| Statistic | 5 | 50 | 230 | 1000 |
| mean | 0.0109 | 0.0109 | 0.0111 | 0.0111 |
| standard deviation | 0.0286 | 0.0092 | 0.0044 | 0.0021 |
| quartile 1 | 0.0007 | 0.0043 | 0.0079 | 0.0096 |
| median | 0.0022 | 0.0076 | 0.0105 | 0.0109 |
| quartile 3 | 0.0072 | 0.0148 | 0.0137 | 0.0124 |
| lower confidence limit | 0.0001 | 0.0018 | 0.0044 | 0.0074 |
| upper confidence limit | 0.0978 | 0.0345 | 0.0213 | 0.0155 |
| Pr(2×) | 0.7774 | 0.5671 | 0.2274 | 0.0105 |
| Pr(4×) | 0.5701 | 0.2483 | 0.0153 | <0.0001 |

results is presented in Table 3.7. Most of the statistics are self-explanatory. Then mean represents the mean of the 10 000 means in $\bar{\mathbf{b}}$. The standard deviation is the standard deviation of $\bar{\mathbf{b}}$. The 95% confidence limits are the rates that correspond to the 2.5th and 97.5th percentiles of the data in $\bar{\mathbf{b}}$. Any rate that falls within these confidence limits is deemed to be not significantly different from the sample mean.

The statistics labeled Pr(2×) and Pr(4×) require further explanation. They were developed in response to the observation that Murphy's estimate of the natural disturbance rate is four times greater than Cumming's. They represent the probability that the mean burn rates calculated for any non-intersecting pair of $T$-year sample periods differ by at least a multiplicative factor of two or four, respectively. The statistics were developed from the elements of $\bar{\mathbf{b}}$ as follows. Let $\mathbf{p}$ be a $D \times 1$ vector representing the ratio of the larger of two adjacent elements in $\bar{\mathbf{b}}$ to the smaller. The elements of $\mathbf{p}$ are calculated as

$$p_1 = \max(\bar{b}_1, \bar{b}_D)/\min(\bar{b}_1, \bar{b}_D) \quad (3.12)$$

$$p_d = \max(\bar{b}_d, \bar{b}_{d-1})/\min(\bar{b}_d, \bar{b}_{d-1}), \quad d = 2,3,\ldots,D. \quad (3.13)$$

Pr(2×) and Pr(4×) are calculated by determining the proportion of the elements in $\mathbf{p}$ that are greater than two or four, respectively. Because annual area burned is assumed to be serially independent, these statistics apply to any pair of non-intersecting $T$-year sample periods, not just adjacent pairs.

The mean burn rates presented in Table 3.7 ($\sim$ 1.1%/year) are quite different than the mean rate calculated earlier for the unconstrained lognormal distribution (1.78%/year). This difference occurs because the burn rate in the simulations is constrained to be at most 100% of the area. Based on the CDF for the distribution, one would expect more than 100% of the area to burn in 0.23% of random draws. These infrequent, but large, outcomes contribute greatly to the calculated mean. Constraining these outcomes to be 100% of the area or less accounts for the difference between the simulated means and that calculated for the distribution. The large difference between the estimates of the mean and median for sample periods of 5 and 50 years is an indication that the distribution of burned areas is highly skewed. The pre-suppression mean burn rate of 0.56% ($47\,967.66/(8.6 \times 10^6) = 0.0056$) from the 35 years of data in Table 3.1 is well within the 95% confidence limits for a sample period of 50 years.

The most important information in Table 3.7 relates to the variability of the estimates of mean annual burn rate. Both 0.5% and 2% are within the confidence limits for sample periods of up to 230 years. Based on the simulation results for a 50 year sample period, one could expect the mean rates calculated for two non-intersecting sample periods to differ by a multiplicative factor of at least 4, approximately one draw in every four.

Estimates of the mean annual rate of burn under the lognormal disturbance regime are highly variable even with sample period as long as 230 years. The variability is so great that estimates as low as 0.5% and as high as 2.0% cannot be statistically differentiated from the sample mean ($\sim$ 1.1%).

## 3.4 Age Class Distributions

The previous section showed that estimates of the annual average burn rate for the study area are imprecise even with very long sample periods. Determination of the "correct" disturbance rate to emulate under NDM management is unlikely given the variability in annual area burned and the large influence of the upper tail of the distribution on estimates of the mean. This section presents simulation results that show that attempting to characterize this forest with an equilibrium age class distribution is also unlikely to be successful.

Monte Carlo simulation is used to track the development of 100 simulated forests subject to the lognormal disturbance regime over a period of 1000 years. This time period should be long enough to allow the age class structure to reach an equilibrium, if one exists. The model is aspatial and the probability of burning is independent of stand characteristics. The starting point for each forest is a fully regulated forest with a maximum age of 99 years (*i.e.* an equal area of the forest is in each one-year wide age class between 0 and 99 years). The regulated forest serves only as a convenient starting point for the simulations and has little or no effect on the ending age distribution. The sample period is long enough that only a very small proportion of the forest area remains unburned after 1000 years.

For each of the simulated forests, the following steps are taken in the simulation. Initialize a $100 \times 1$ age class vector $\mathbf{g_0}$ whose elements $g_{i0}$ represent the initial age (years) of each of the age classes $i$ in the forest:

$$g_{i0} = i - 1, \quad i = 1, 2, \ldots, 100. \tag{3.14}$$

Initialize a $100 \times 1$ area proportion vector $\mathbf{a}_0$ whose elements $a_{i0}$ represent the proportion of forest area in each age class. Because the starting point is a regulated forest, an equal proportional area is assigned to each age class:

$$a_{i0} = 0.01, \quad i = 1, 2, \ldots, 100. \tag{3.15}$$

Create a $1000 \times 1$ vector $\lambda$ containing the annual burn rate for each year of the simulation. The burn rate for each year $\lambda_t$ is randomly drawn from a constrained lognormal distribution:

$$\lambda_t = \min(1, \exp x), \quad x \sim N(\mu, \sigma^2), \quad t = 1, 2, \ldots, 1000. \tag{3.16}$$

Again, these simulations use $\mu = -8.096$ and $\sigma = 2.853$ as determined earlier. For each each year in the simulation, a new age class vector $\mathbf{g}_t$ is created by incrementing the elements of $\mathbf{g}_{t-1}$ by one and joining a zero to the result. A new proportional area vector $\mathbf{a}_t$ is created by proportionally reducing the elements of $\mathbf{a}_{t-1}$ to reflect the burn rate, and joining the burn rate to the result.

$$\mathbf{g}_t = \begin{bmatrix} 0 \\ 1 + \mathbf{g}_{t-1} \end{bmatrix}, \quad \mathbf{a}_t = \begin{bmatrix} \lambda_t \\ (1 - \lambda_t)\mathbf{a}_{t-1} \end{bmatrix}, \quad t = 1, 2, \ldots, 1000. \tag{3.17}$$

The ending age class distribution for the simulated forest is defined by the vectors $\mathbf{g}_{1000}$ and $\mathbf{a}_{1000}$.

The ending age class distributions for the first eight forests generated by the simulation are presented in Figure 3.7. The proportional areas in one-year wide age classes from $\mathbf{a}_{1000}$ were aggregated into 20-year wide age classes for presentation purposes. Age class 190 is a collector age class representing the proportional area of the forest 180 years of age or older. No regular pattern is visually apparent in these outcomes. None of these age class distributions resembles a negative exponential distribution.

No firm conclusion about age class distributions should be drawn solely on the basis of visual inspection of eight outcomes of a simulation model. Figure 3.8 provides an indication of the variability in the ending age class structure of the forest. The figure summarizes the distribution of proportion of area in 20-year wide age classes at end of the simulation period by plotting the mean, range, quartiles, and the upper and lower eighths of the simulation results.

The variability in simulated age class distributions after 1000 years is extreme, although there is evidence of a downward trend in proportional area occupied by increasing age class. This variability occurs because relatively infrequent, extreme fire years produce spikes in the age class distribution of the forest. These spikes grow older

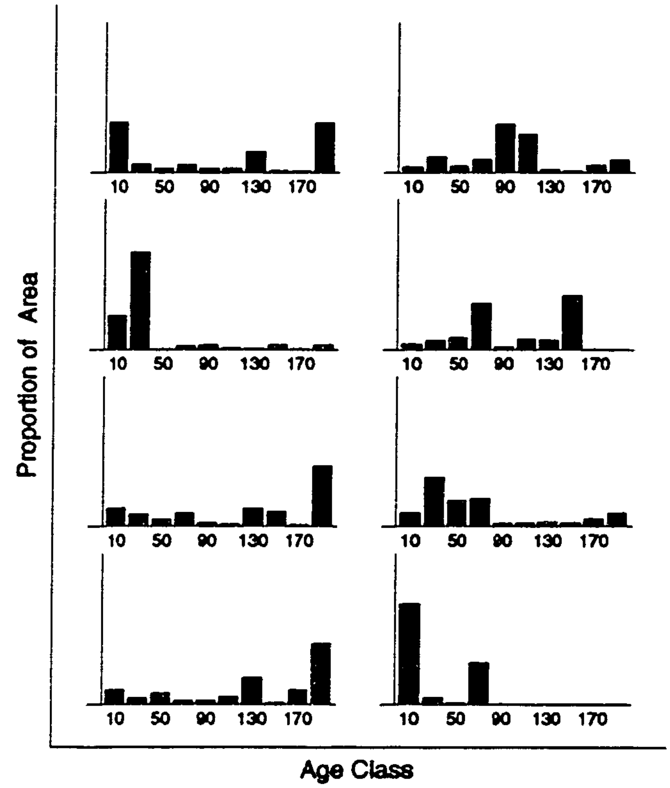


Figure 3.7: First eight age class distributions resulting from Monte Carlo simulations of the lognormal disturbance regime. The length of the y-axis in each panel represents 100% of the area.

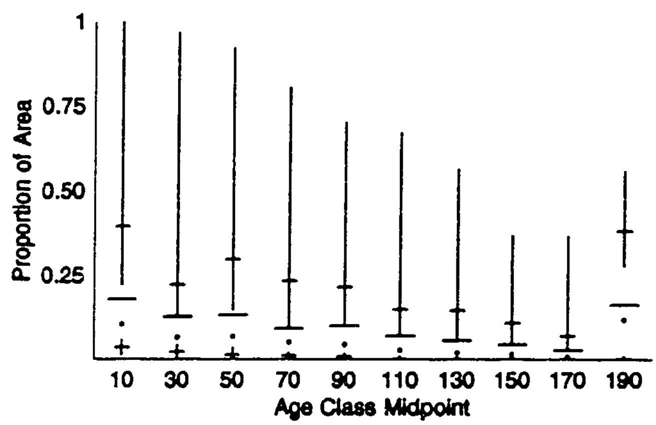


Figure 3.8: Summary of simulated ending age class distributions. The median proportion of area occupied by each age class is represented by the dot. The other quartiles are indicated by the inner endpoints of the whiskers. The minimum and maximum are indicated by the outer endpoints of the whiskers. The mean is indicated by the wide dash. The upper and lower eighths are indicated by the narrow dashes. Age class 190 is a "collector" representing all ages 180 years or greater.

and move to the the right on the age class distribution plot until another major fire year substantially reduces the size of the spike.

Van Wagner (1978) recognized that fluctuations in annual area burned could affect the validity of his age class distribution model. He states

> ... there is no reason why burned area could not vary from year to year as long as the long-term average was constant and the time scale of the fluctuations small compared with the length of the fire cycle. Such departures from the ideal would naturally result in statistical roughness but need not disqualify the negative exponential concept.

For the study area, the variability in annual area burned is large enough to disqualify the negative exponential concept. The simulations show that achieving such an age class distribution is highly unlikely. I suspect that it is also true for areas of similar size in most of the boreal forest.

There appears to be no equilibrium age class distribution for the study area. This is consistent with results from other research (Baker, 1989a,b; Cumming *et al.*, 1996; Romme, 1982). The system may be in a stochastic equilibrium in the sense that the two parameters of the lognormal disturbance regime could be relatively stable, even though the resulting age class distribution is highly unstable. However, the time series of data used here is too short to confidently characterize the disturbance regime as stationary.

It is of interest to note that the age class distribution that results from averaging the results of the 100 simulations is very close to a negative exponential distribution. However, this does not affect the conclusions drawn from this study. It would be relevant if the annual probabilities of fire at different points in the forest were independent of each other.

## 3.5 Conclusions and Implications

The natural disturbance regime for the study area is characterized by large annual variations in the annual area burned. The distribution of annual area burned or burn rates is well-described by a lognormal distribution. The area burned in any one year can be modeled as an independent random draw from a lognormal distribution, constrained such that the area burned never exceeds the total area. This characterisation may also apply to other areas of the boreal mixedwood forest.

Estimates of average burn rates for the area developed through Monte Carlo simulation are highly imprecise even with very long sample periods (despite the fact that the mean of the natural logarithm of disturbance rate can be estimated precisely). It appears that nature will not provide us with a characterisation of the mean rate of natural disturbance precise enough to dictate harvest rate decisions under NDM management. The two-parameter characterisation of the natural disturbance regime presented here could, however, be valuable for quantification of the risks associated with forest management and evaluation the effectiveness of fire suppression. Forest managers will make the harvest rate decision considering whatever factors are important to them, but defaulting to an unquantifiable natural disturbance rate is not an option.

The disturbance regime modeled here does not lead to an equilibrium age class distribution. Any claim that the natural age class distribution for the study area follows a particular functional form (*e.g.* negative exponential or Weibull functions) is highly questionable. Therefore, specifying a target age class distribution as a management goal on the basis of "naturalness" is inappropriate.

Many studies have attributed apparent changes in rates of disturbance to changes in climate or fire suppression efforts on the basis of an analysis of the age class structure of a forest (*e.g.* Murphy, 1985; McCune, 1983; Johnson *et al.*, 1995; Turner and Romme, 1994). I suggest that, at least in forests with fire regimes as variable as in the study area, many of these observed changes may be more parsimoniously attributed to random chance than to climate change or anthropogenic effects.

There is, of course, substantial room for further research. The applicability of this model to other study areas should be examined. The lognormal characterisation of annual burn rate could provide a useful basis for evaluating fire risk, and in evaluating the effectiveness of fire suppression. This fire risk model could be incorporated into forest planning systems to better evaluate the timber supply and financial and ecological risks associated with forest fire. The models developed here could be improved by incorporating the spatial aspects of forest fire, and recognizing that different parts of a forest (perhaps classified by species composition, age, soil moisture regime, *etc.*) may face different fire regimes.

# Bibliography


Alberta Pacific Forest Industries Inc. undated. Environmental management. [web page]. <http://www.alpac.ca/regional.html> [Accessed 11 June 1998].

Baker W.L. 1989a. Effect of scale and spatial heterogeneity on fire-interval distribution. Can. J. For. Res. **19**: 700–706.

Baker W.L. 1989b. Landscape ecology and nature reserve design in the Boundary Waters Canoe Area, Minnesota. Ecology **70**: 23–35.

Boychuk D. and Perera A.H. 1997. Modeling temporal variability of boreal landscape age-classes under different fire disturbance regimes and spatial scales. Can. J. For. Res. **27**: 1083–1094.

Cumming S.G. 1997. Landscape dynamics of the boreal mixedwood forest. Ph.D. dissertation, Department of Forest Sciences. University of British Columbia, Vancouver.

Cumming S.G., Burton P.J., and Klinkenberg B. 1996. Boreal mixedwood forests may have no "representative" areas: Some implications for reserve design. Ecography **19**: 162–180.

Davis C.S. and Stephens M.A. 1989. Empirical distribution functions and goodness-of-fit tests. Applied Statistics **38**: 535–543. Algorithm AS 248.

Government of Alberta. 1997. Alberta fire incidence data. [web page]. <http://www.gov.ab.ca/~env/lfs/fpd/firedata/firedata.html> [Accessed 24 June 1998].

Harvey A.C. 1981. The Econometric Analysis of Time Series. Philip Allan, Oxford.

Heinselman M.L. 1973. Fire in the virgin forests of the Boundary Waters Canoe Area, Minnesota. Quaternary Research **3**: 329–382.

Ingersoll Jr. J.E. 1987. Theory of Financial Decision Making. Rowman & Littlefield, Savage, MD.

Johnson E.A. and Gutsell S.L. 1994. Fire frequency models, methods, and interpretations. In *Advances in Ecological Research* (Edited by M. Begon and A.H. Fitter), vol. 25. Academic Press, London, 239–287.

Johnson E.A., Miyanishi K., and Weir J.M.H. 1995. Old-growth, disturbance, and ecosystem management. Canadian Journal of Botany **73**: 918–926.

Kabzems A., Kosowan A.L., and Harris W.C. 1976. Mixedwood section in an ecological perspective. Technical Bulletin 8, Saskatchewan Parks and Renewable Resources, Forestry Division, Regina, Saskatchewan.

McCune B. 1983. Fire frequency reduced two orders of magnitude in the Bitterroot Canyons, Montana. Can. J. For. Res. **13**: 212–218.

Murphy P.J. 1985. Methods for evaluating the effects of forest fire management in Alberta. Ph.D. dissertation, Department of Forestry. University of British Columbia, Vancouver.

Romme W.H. 1982. Fire and landscape diversity in subalpine forests of Yellowstone National Park. Ecological Monographs **52**: 199–221.

Rowe J.S. 1972. Forest Regions of Canada. Canadian Forestry Service, Department of Fisheries and the Environment, Government of Canada, Ottawa. Publication No. 1300.

Sokal R.R. and Rohlf F.J. 1981. Biometry. W.H. Freeman and Company, San Franciso, 2nd edn.

Stephens M.A. 1986. Tests based on EDF statistics. In *Goodness-of-Fit Techniques* (Edited by R.B. D'Agostino and M.A. Stephens), chap. 4. Marcel Dekker, Inc., New York.

Turner M.G. and Romme W.H. 1994. Landscape dynamics in crown fire ecosystems. Landscape Ecology **9**: 59–77.

Van Wagner C.E. 1978. Age-class distribution and the forest fire cycle. Can. J. For. Res. **8**: 220–227.

Weyerhaeuser Canada. 1997. Twenty-year forest management plan & environmental impact statement: Summary. [web page]. <http://www.link.ca/~20year/execsumm/3.htm> [Accessed 11 June 1998].

White K.J. 1997. SHAZAM User's Reference Manual Version 8.0. McGraw-Hill.

# Chapter 4

# Financial, Timber Supply, and Wildlife Habitat Trade-Offs in a Stochastic Environment

## 4.1 Introduction

The characterization of the natural disturbance regime (NDR) presented in Chapter 3 and Armstrong (1999) has implications for natural disturbance based approaches for forest management. The interannual variability in disturbance rates is an important characteristic of the NDR. The relatively infrequent occurrence of extreme fire years largely determines the age class structure of the forest. The mean rate of natural disturbance is not precisely quantifiable. Therefore, attempting to approximate the natural rate of disturbance through management practices is untenable. This also means that other single parameter characterizations of the NDR for the study area(*e.g.* fire cycle or fire return interval) are not meaningful. There is no equilibrium age class distribution for a forest subject to this NDR: management regimes designed to lead to a particular "natural" forest age class distribution are misguided. For example, the negative exponential age class distribution model proposed by Van Wagner (1978) almost never holds for a forest subject to the lognormal disturbance regime. The negative exponential age class distribution is not necessarily an invalid goal for the future age class structure of a forest. However, rationalizing this goal on the basis that it is the equilibrium structure for the forest is incorrect.

This is not to say that the natural disturbance model of forest management is inappropriate for the boreal mixedwood forest. The single rate approach to NDM is invalid, as is a target age class distribution. There are, however, alternative interpretations. This chapter presents an approach to the determination of appropriate harvest levels using the natural variability in forest structure as a guide.

A highly aggregated aspatial representation of the forest is used in the models presented here. This is a departure from the current trend in Canada towards very detailed spatially explicit representations of forests in planning models. There are a number of reasons for my choice of an aggregated aspatial representation.

1. There is a limited understanding of the biophysical relationships between neighboring forest stands, and the wildlife species that inhabit them. In many cases, the level of detail tracked by spatially explicit models is not warranted by the questions being asked of the models, or by the level of understanding of inter-stand relationships. The questions being asked in this study do not require a spatial model.

2. Highly detailed, spatially explicit models of large forest areas are very expensive to run in terms of time and computing power. There is a trade-off between the number of runs that can be made and the level of spatial detail tracked. This limits the number of alternative scenarios that can be examined. This makes it more difficult to recognize forest level patterns and implement Monte Carlo analysis.

3. Optimization of forest activity schedules in a spatially explicit framework is difficult. The types of spatial optimization models described by Hof and Bevers (1998) work for very small problems, but are inappropriate for forests comprised of several thousand stands. Optimization models are more appropriate than simulation models for analyses of trade-offs between competing values.

4. Most importantly, for a system subject to a stochastic disturbance regime (*e.g.* the boreal mixedwood forest), any projection of a spatially explicit model will represent just one possible outcome of the stochastic process. Accurately predicting the spatial location and extent of the forest fires that will occur 50 years in the future, or even next year, is impossible, or at least very difficult. In a system such as this, a statistical characterization of the range of possible outcomes will be more useful than one spatially explicit projection of one possible outcome. The large number of runs required to characterize the distribution of outcomes for a large forest area is likely to be prohibitively expensive with a high resolution, spatially explicit model.

In summary, the speed of analysis that is possible with a highly aggregated aspatial representation of the forest allows for a better understanding of the range of possible outcomes of a highly stochastic system. Spatially explicit models are useful tools for many types of analysis. However, there are some questions more appropriately addressed with aggregated aspatial models.

Monte Carlo simulation is used to project the distribution of the structure of a forest (defined in terms of area in age class by cover type combinations) subject to the lognormal NDR. The age class structure is summarized as the empirical probability density function (EDF) of the area of habitat available for five vertebrate species in each year of the projection period. Habitat is defined in terms of cover group and age ranges. This proves to be a convenient shorthand representation of the structure of the forest. The simulation model actually tracks the structure of the forest in terms of the area in 1-year wide age class in each cover type. It would be possible to determine EDFs for each of these combinations. However, it would be difficult to make sense of 1000 projected EDFs (200 years by 5 cover groups). Aggregating this information in terms of distributions of habitat area for a handful of species allows for easier interpretation of the results.

The projected EDFs for each period are described using quantiles. The quantile $x$, where $x$ is a real number between 0 and 1, is expressed as the area of habitat $y$, where the proportion $x$ of the simulation realizations are less than $y$ ha. These quantiles apply to each habitat level, for each species, and for each time step of the simulation. The creation of these quantiles is discussed in Section 4.3.

A linear programming based optimization model is used to quantify the trade-offs between financial objectives of forest management and habitat areas for each of the five vertebrate species. The objective of the model is to maximize the net present value (NPV) of forest management activities. The projected EDFs for habitat areas are used to set periodic constraints on wildlife habitat. Several optimization runs are made, with habitat area constraints set to reflect different quantiles from the EDFs. For example, in one run the habitat areas for each of the five species are simultaneously constrained to be at least the 0.10 quantile from the projected EDFs. In another run, habitat areas are constrained to be at least the 0.40 quantile. By repeating this process for several quantile levels, a trade-off curve between net present value and habitat availability quantiles can be developed. In one sense, the most important decision required for a forest manager is the desired position on this trade-off curve.

The remainder of this chapter is organized as follows. The next section describes the input data used for the simulation and optimization runs. This is followed by a section describing the simulation model and the results of the Monte Carlo simulations. The optimization model and the construction of the trade-off curves are then presented. The chapter concludes with a summary discussion and suggestions for further research.

## 4.2 Input Data

The inventory and yield relationships used in this study are based on data provided by Daishowa-Marubeni International Ltd. (DMI) for part of their Forest Management Agreement Area in north-central Alberta, Canada. The area is an important timber producing area for a pulp mill and several sawmills. The study area is approximately bounded by 56°N and 57°40′N latitude and 115°W and 117°W longitude. The starting inventory is shown in Table 4.1. This inventory represents the current condition of the 888 713 ha of net merchantable land base for the study area. The net merchantable land base is the part of the total forest area considered available for timber harvest activities. The area is net of stands which are considered never merchantable due to low projected volume, muskegs and and other wetlands, and areas deleted for stream and lake buffers,

Table 4.1: Starting inventory for study area. Area (ha) by age and cover type

| Age (years) | White spruce | Aspen | Mixed | Pine | Black spruce | Total |
|---|---|---|---|---|---|---|
| 10 | 2 | 5 | | | 18 | 25 |
| 20 | 70 | 636 | 2 483 | 6 210 | 521 | 9 920 |
| 30 | 5 | 3 400 | 710 | 231 | 3 | 4 349 |
| 40 | 1 050 | 1 304 | 1 049 | 663 | 128 | 4 194 |
| 50 | 4 552 | 61 422 | 7 196 | 2 402 | 2 068 | 77 640 |
| 60 | 18 970 | 224 645 | 33 004 | 31 950 | 15 674 | 324 243 |
| 70 | 8 420 | 82 523 | 11 718 | 10 675 | 11 164 | 124 500 |
| 80 | 7 307 | 39 726 | 8 289 | 3 743 | 6 805 | 65 870 |
| 90 | 6 531 | 11 763 | 6 203 | 2 364 | 11 545 | 38 406 |
| 100 | 14 407 | 30 753 | 12 688 | 3 844 | 12 275 | 73 967 |
| 110 | 8 310 | 11 674 | 5 386 | 2 425 | 4 686 | 32 481 |
| 120 | 11 015 | 12 301 | 9 348 | 1 144 | 2 534 | 36 342 |
| 130 | 17 193 | 7 554 | 8 802 | 330 | 4 516 | 38 395 |
| 140 | 17 398 | 8 052 | 6 309 | 426 | 4 027 | 36 212 |
| 150 | 6 779 | 1 590 | 3 005 | 695 | 2 498 | 14 567 |
| 160 | 1 614 | 198 | 243 | 912 | 1 781 | 4 748 |
| 170 | 443 | 25 | 12 | | 73 | 553 |
| 180 | 23 | | | | 20 | 43 |
| 190 | 384 | | 134 | 6 | 289 | 813 |
| 200 | | | 26 | | 22 | 48 |
| 210 | 90 | | | | 265 | 355 |
| 260 | | | | | 42 | 42 |
| Total | 124 563 | 497 571 | 116 605 | 68 020 | 80 954 | 887 713 |

and other operational considerations. Softwood volumes are assumed to change with stand age according to the yield tables presented in Figure 4.1. Hardwood volumes develop as shown in Figure 4.2.

The most striking feature of the starting inventory presented in Table 4.1 is the large area of forest in the 60 year age class (324 243 ha or 36.5% of the land base). This spike in the age class distribution is characteristic of forests subject to the lognormal disturbance regime characterized in Chapter 3. Most of the yield curves presented in Figures 4.1 and 4.2 show declining volumes somewhere between 100 and 150 years of age. These declines reflect stand break-up.

Cumming *et al.* (1994) used a deterministic simulation framework to project the development of part of the boreal mixedwood to examine potential conflicts between wildlife habitat and timber supply. Their representation of habitat quality is used for the current study. Cumming *et al.* (1994) described the forest in terms of area in cover type by habitat stage combinations. The cover types were based on species composition of stands. The recognized cover types were pine, white spruce, aspen, mixed, and black spruce. The white spruce, aspen, and mixed cover types represent stands that occur on

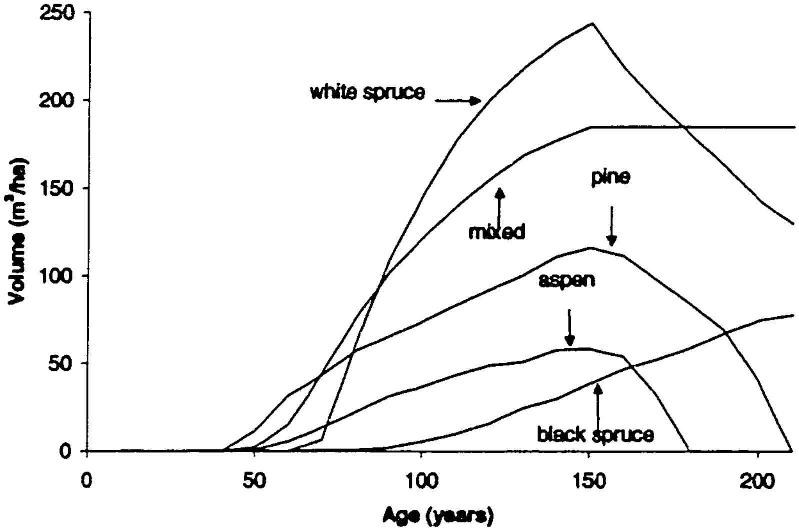


Figure 4.1: Softwood volume yields (m$^3$ ha$^{-1}$ by cover type and age (years)).

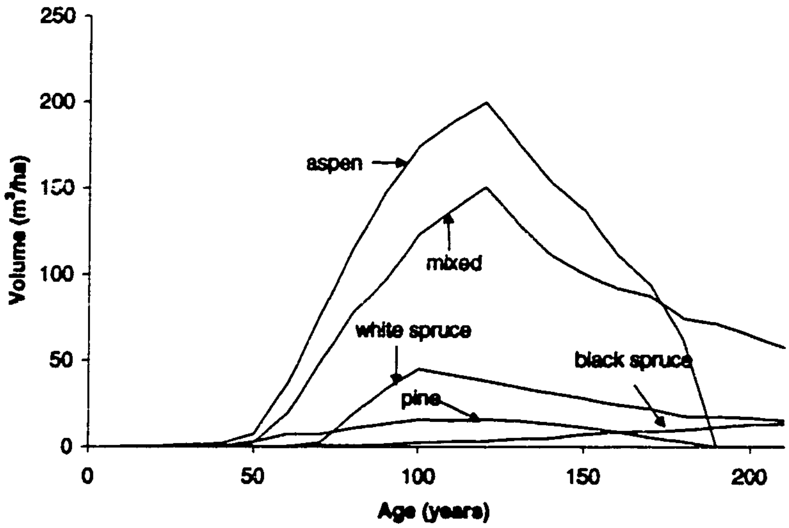


Figure 4.2: Hardwood volume yields (m$^3$ ha$^{-1}$ by cover type and age (years)).

Table 4.2: Habitat stage definition by cover type and age range. After Cumming *et al.* (1994).

| Habitat stage | | Age range (years) | |
|---|---|---|---|
| ID | Description | Aspen cover type | Other cover types |
| 1 | Establishment | 0 – 5 | 0 – 5 |
| 2 | To maximum stem density | 6 – 15 | 6 – 25 |
| 3 | To maximum crown closure | 16 – 30 | 26 – 60 |
| 4 | To maximum basal area | 31 – 60 | 61 – 100 |
| 5 | Maturity | 61 – 80 | 101 – 150 |
| 6 | Overmaturity | 81+ | 151+ |

mesic sites. They are differentiated based on crown closure by species group. The white spruce group comprises mesic stands with 80% or more of the crown area occupied by softwood species; the aspen group comprises mesic stands with 80% or more of the crown area occupied by hardwood species; and the mixed group represents all other mesic stands. Six habitat stages were recognized: establishment, the interval to maximum stem density, the interval to maximum crown closure, the interval to maximum basal area, a mature stage, and an overmature stage. These habitat stages were related by Cumming *et al.* (1994) to stand age and cover type as shown in Table 4.2.

Cumming *et al.* (1994) relate cover type and habitat stage to habitat quality for five vertebrate species: pine marten (*Martes americana* Turton), meadow vole (*Microtus pennsylvanicus* Ord), broad-winged hawk (*Buteo platypterus* Vieillot), black-throated green warbler (*Dendroica virens* Gmelin), and northern three-toed woodpecker (*Picoides tridactylus* Linnaeus). These species were selected because of the diversity of their habitat requirements and because of their relatively small ranges which better allows for representation of habitat quality in an aspatial model (as opposed to large mammals which may require a spatially arranged diversity of habitat types). Cumming *et al.* (1994) justify their choice of species as follows:

> The pine marten was chosen for its preference for mature stands containing white spruce, and because it and other large mustelids face habitat losses throughout the circumpolar boreal forests .... The meadow vole illustrates a species dependent upon open, recently disturbed habitat. The tree-toed woodpecker is characteristic of old coniferous stands, whereas the black-throated green warbler is associated with mature and older mixed and coniferous stands. The broad-winged hawk nests and forages almost exclusively in mature deciduous stands.

For the remainder of this chapter, the vertebrate species will be referred to by the following abbreviations: PIMA (pine marten), MEVO (meadow vole), BWHK (broad-winged hawk), TTWP (three-toed woodpecker), and BTGW (black-throated green warbler).

The habitat quality indices developed by Cumming *et al.* (1994) and presented in Table 4.3 were based largely on a literature review. Habitat quality index is coded as an integer between one and six inclusive, where one represents unsuitable habitat and six represents ideal habitat. This numeric scale follows that used by McNichol *et al.* (1981) to indicate avian abundance in different habitats. It proves to be convenient for the modeling presented here.

Table 4.3: Habitat quality index by vertebrate species, cover type, and habitat stage. Blank entries represent a habitat quality index of 1. Reproduced from Cumming *et al.* (1994), Table 3.

| | | Habitat stage | | | | | |
|---|---|---|---|---|---|---|---|
| Species | Cover type | 1 | 2 | 3 | 4 | 5 | 6 |
| PIMA | Pine | | | 2 | 2 | 2 | 2 |
| | White spruce | | | 2 | 3 | 4 | 6 |
| | Mixed | | | | 2 | 3 | 4 |
| MEVO | Pine | 3 | 2 | | | | |
| | White spruce | 6 | 3 | | | | |
| | Aspen | 6 | 3 | | | | |
| | Mixed | 6 | 3 | | | | |
| | Black spruce | 3 | 2 | | | | |
| BWHK | Aspen | | | | | 4 | 6 |
| | Mixed | | | | 4 | 5 | 4 |
| TTWP | Pine | 4 | | | 2 | 4 | 5 |
| | White spruce | | | | 3 | 4 | 6 |
| | Mixed | | | | 2 | 3 | 4 |
| | Black spruce | 3 | | | 2 | 4 | 6 |
| BTGW | White spruce | | | 2 | 4 | 5 | 4 |
| | Mixed | | | 2 | 4 | 6 | 6 |

## 4.3 Monte Carlo Simulation

Monte Carlo simulation is a technique used when one or more of the variables in the simulated system is a random variable. The value that such a variable takes on in a simulation is determined by a random draw from the variable's assumed probability density function. By repeating the simulation procedure several times, one can develop an understanding of the probability density functions for outputs of the modeled system.

Monte Carlo simulations were used to project the probability distribution of habitat areas for each of the five vertebrate species. One thousand simulated inventory projections were run, each of which projected the development of the forest for 100 years. The starting point of each projection was taken to be the current inventory (Table 4.1). In each year of each simulation, the annual burn rate ($\lambda_t$) was drawn from the lognormal distribution identified by Armstrong (1999).

$$\lambda_t = \min(0.20, \exp x), \quad x \sim N(\mu, \sigma^2), \quad t = 1, 2, \ldots, 100. \tag{4.1}$$

These simulations use $\mu = -8.096$ and $\sigma = 2.853$ as determined in Chapter 3. These parameters are easily interpretable: $\mu$ is the mean of the natural logarithm of the annual proportion of the area burned under a natural disturbance regime; $\sigma$ is the standard deviation. The annual proportion of area burned for this study was truncated at 0.20 in order to prevent burn proportions much greater than evident from the historical record.

The simulation model takes the following steps:

1. For each of the 1000 simulation runs
   (a) Retrieve the initial inventory as area in cover type by age combinations.
   (b) For each of the 100 years in the simulation
      i. Randomly draw the annual disturbance rate from the distribution shown in Equation 4.1.
      ii. Determine the area burned in each cover type by age cell by multiplying the area of the cell by $\lambda_t$. Set the age of the burned proportion to zero. Increment the age of the remainder by one year.
      iii. Use Table 4.2 to assign a habitat stage to each new cover type by age cell.
      iv. Use Table 4.3 to assign a habitat quality index to each cover type by age cell for each species.
      v. For each of the habitat quality indices from 2 through 6
         A. Calculate the total area of habitat of at least the habitat quality index being process
         B. Store the results for later processing.
2. For each of the 100 years in the simulation
   (a) For each of the five vertebrate species
      i. For each of the habitat quality indices from 2 through 6
         A. Sort the stored results by ascending total area.
         B. Assign a quantile to each item in the sorted list (position in the list divided by 1000)
         C. Store the results

In each year, each cover type by age combination was assigned a habitat stage according to Table 4.2. The projected habitat quantiles under natural disturbance for PIMA, MEVO, BWHK, TTWP, BTGW are shown in Figures 4.3 through 4.7. These simulations represent the development of the forest in the study area over a 100 year period in response to the natural fire regime. No timber harvest or fire protection activities occur in the simulations.

For the remainder of this chapter, the focus is on good habitat, that is, habitat with a quality index of five or greater. The information for good habitat for all five species is duplicated in Figure 4.8 for convenient reference. Figure 4.8 represents the empirical distribution functions of habitat quality used in remainder of this chapter. The age class and and tree species combinations are summarized in terms of area of good habitat for five different vertebrate species, over a period of 100 years.

The habitat area projections for BWHK and for BTGW show a large jump at 20 and 40 years, respectively. This reflects the transition of a large area of aspen from habitat stage 5 to habitat stage 6 (ideal habitat for BWHK) at 20 years from present, and the transition of a large area of mixed from habitat stage 4 mixed to habitat stage 5 at 40 years from present. Both transitions reflect the spike in the age class distribution at 60 years of age. In both cases, the habitat quality index changes from 4 to 6 at the time of the transition.

The probability distributions of the area of good habitat are projected for each of the five species. The panels of the figure show the 95% confidence limits and the quartiles for each of the projections. The main conclusions to draw from this figure are that, under natural conditions, one would expect the median habitat areas for each of the species to change substantially over the projection period, and that variation in projected habitat areas become extremely large in a relatively short period of time (say 20 – 40 years). The large changes in projected median habitat areas indicate that the system is not currently in an equilibrium with respect to area of habitat for the five species examined, and the large variation around the median reflects the non-equilibrium nature of the system as discussed in Chapter 3.

This presents an interesting planning problem. Under the natural disturbance model of management, the goal is to maintain the characteristics of a natural ecosystem. However, the simulations conducted show that there is no one "ecologically correct" mix of habitats for the forest and that the realized mix of habitat areas is likely to change dramatically over time. Another consideration is that there are trade-offs between areas of habitat for the different wildlife species. For example, with the models used here, overmature white spruce is good habitat for PIMA and TTWP, but less than ideal for MEVO, BWHK, and BTGW. Allowing white spruce stands to reach overmaturity delays the creation of new good MEVO, BWHK, and BTGW habitat.

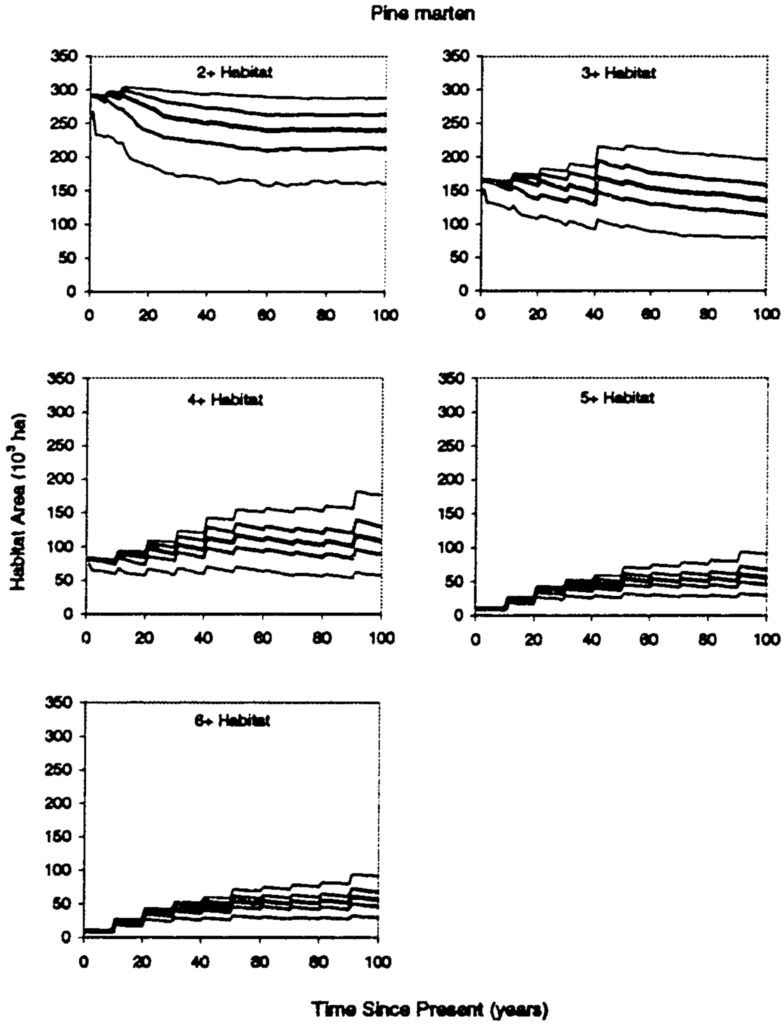


Figure 4.3: Projected habitat quantiles for PIMA. The quantiles shown are 0.025, 0.25, 0.5, 0.75, and 0.975. The panel titles indicate habitat quality (*e.g.* "2+ Habitat" indicates habitat with a quality index of 2 or greater).

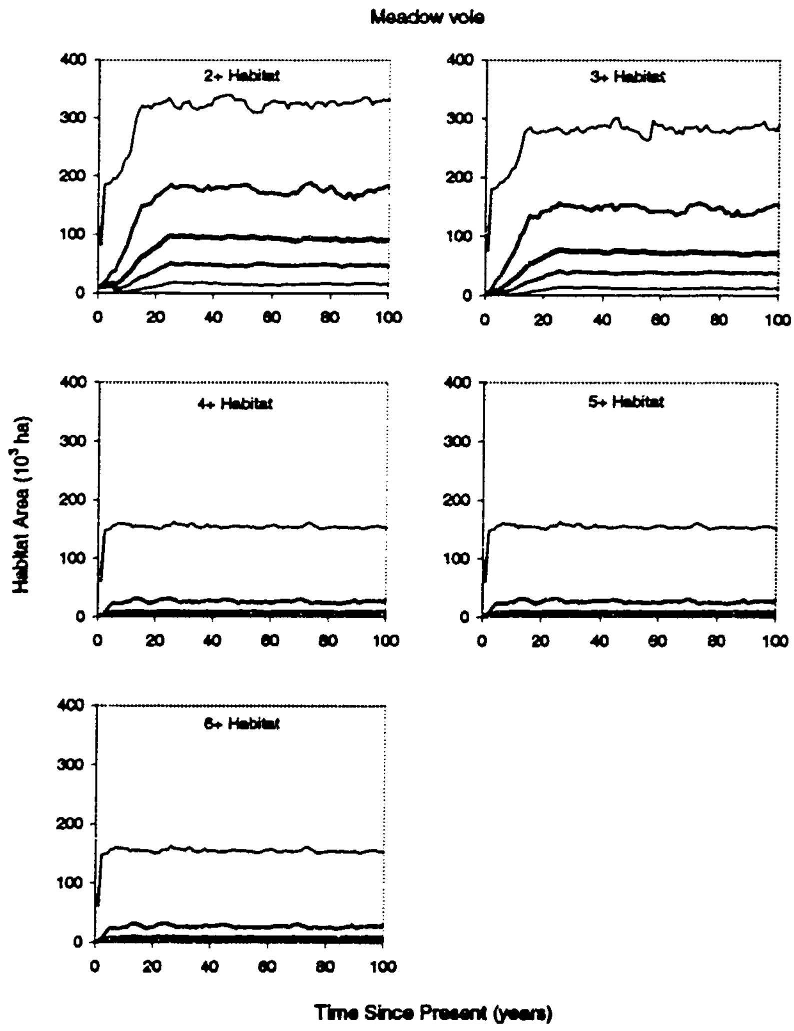


Figure 4.4: Projected habitat quantiles for MEVO. The quantiles shown are 0.025, 0.25, 0.5, 0.75, and 0.975. The panel titles indicate habitat quality (*e.g.* "2+ Habitat" indicates habitat with a quality index of 2 or greater).

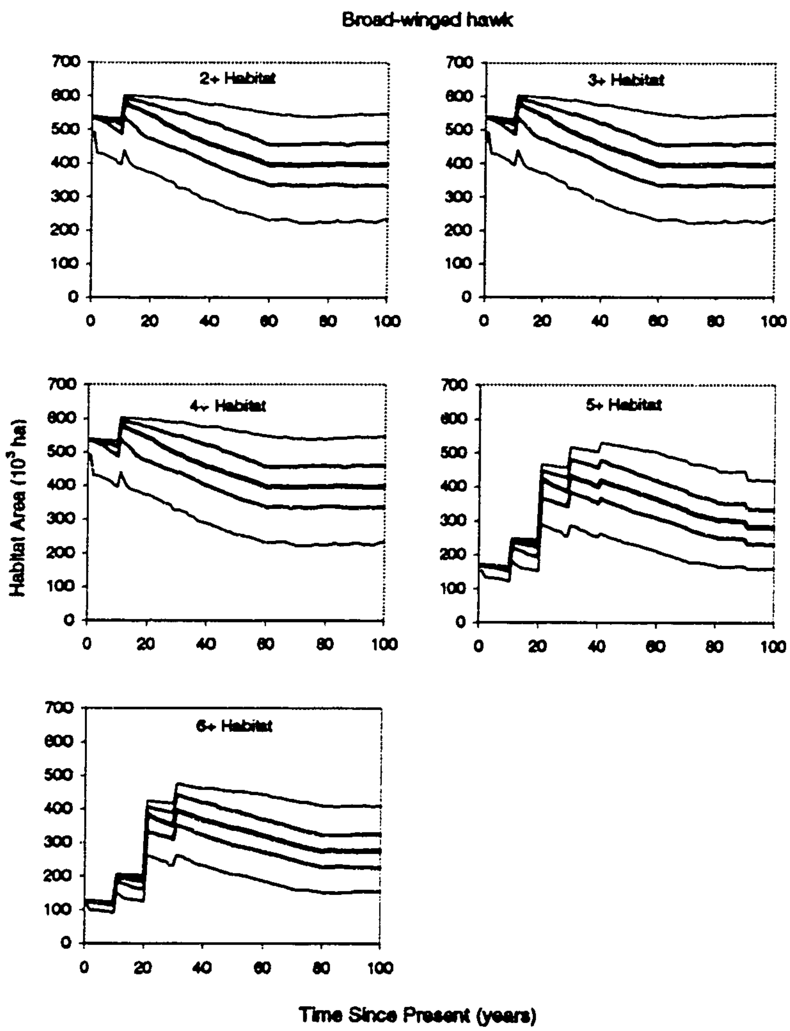


Figure 4.5: Projected habitat quantiles for BWHK. The quantiles shown are 0.025, 0.25, 0.5, 0.75, and 0.975. The panel titles indicate habitat quality (*e.g.* "2+ Habitat" indicates habitat with a quality index of 2 or greater).

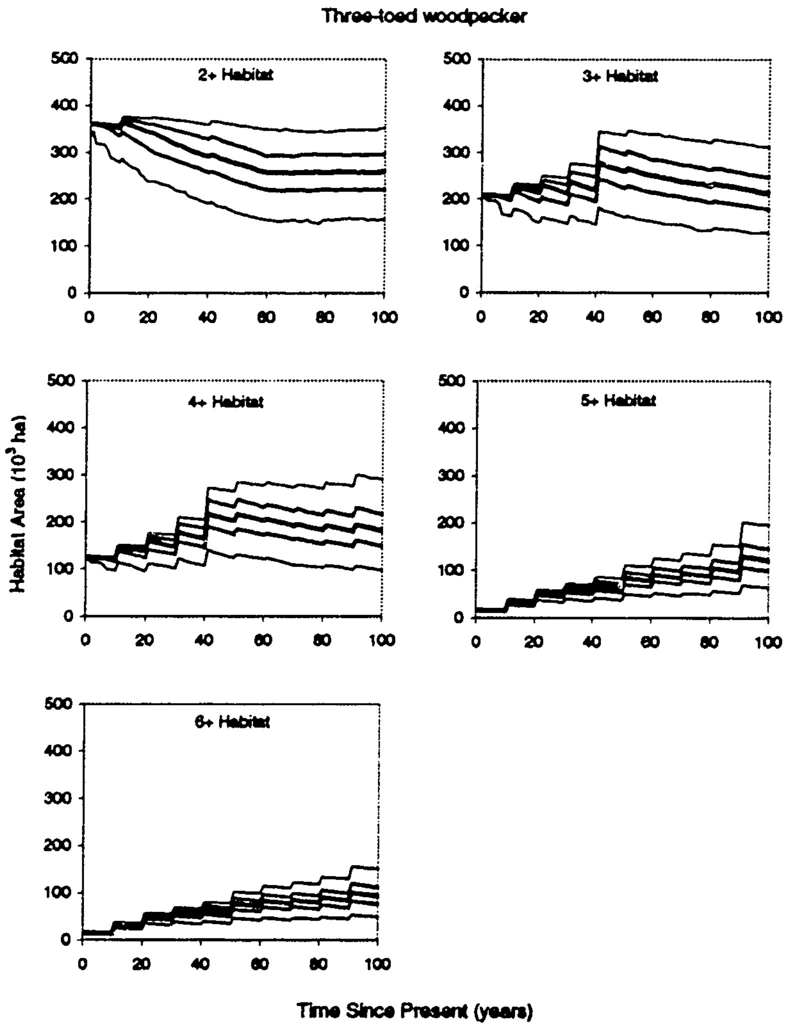


Figure 4.6: Projected habitat quantiles for TTWP. The quantiles shown are 0.025, 0.25, 0.5, 0.75, and 0.975. The panel titles indicate habitat quality (*e.g.* "2+ Habitat" indicates habitat with a quality index of 2 or greater).

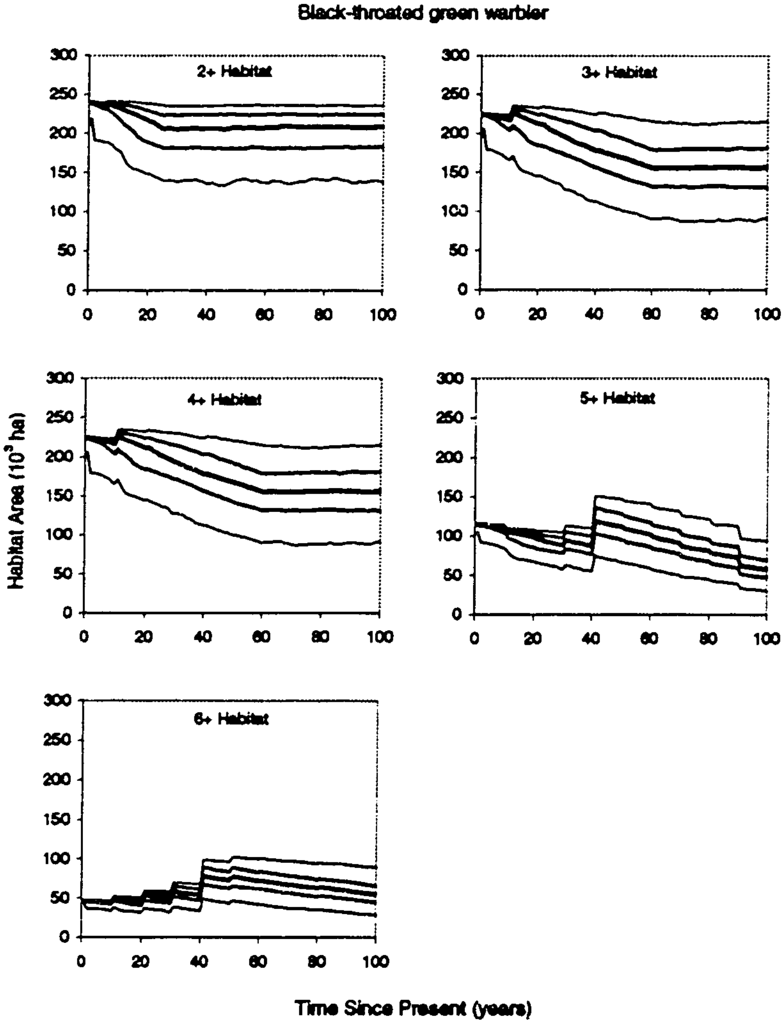


Figure 4.7: Projected habitat quantiles for BTGW. The quantiles shown are 0.025, 0.25, 0.5, 0.75, and 0.975. The panel titles indicate habitat quality (*e.g.* "2+ Habitat" indicates habitat with a quality index of 2 or greater).

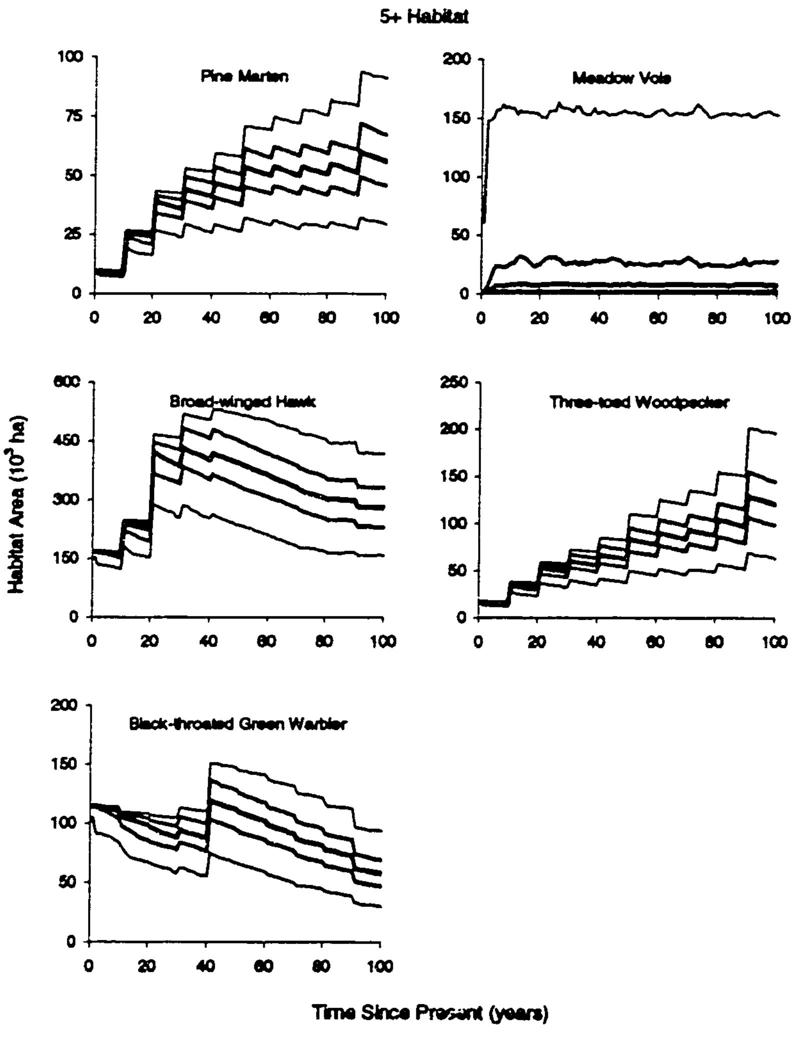


Figure 4.8: Projected habitat area quantiles for habitat quality indices of five or greater for five vertebrates. The quantiles shown are 0.025, 0.25, 0.5, 0.75, and 0.975.

## 4.4 Constrained Optimization

For the purposes of the optimization model, it is assumed that the objective of the forest manager is to maximize the net present value of timber harvest. Fire suppression occurs, and is believed to reduce the occurrence of fire relative to natural conditions, although there is no published evidence to support this belief. The goal of the Alberta Land and Forest Service is to keep the annual area burned to be less than 0.1% of the productive forest area. Assuming that this goal is met, the impact of fire becomes neglible. I ignore the effects of fire in the optimization model.

The annual discount rate is assumed to be 5%. The conversion surplus value is assumed to be 60 \$ $m^{-3}$ for softwood timber and 50 \$ $m^{-3}$ for hardwood timber at the mill gate. As used here, conversion surplus is a measure of the value of logs delivered to the mill. It represents the selling price of the final product (*e.g.* lumber, pulp) less all the variable costs of milling and marketing the product, expressed on a per cubic meter of roundwood basis (Davis and Johnson, 1987). All softwood and hardwood volume is taken from every harvested stand. Stands are assumed to regenerate to the same cover type after harvest. No regeneration lag is modeled. Regeneration costs are assumed to be incorporated into harvest costs. Timber harvest costs are assumed to be 5000 \$ $ha^{-1}$. Non-declining yield constraints are applied to both types of timber: the harvest volume in any one period is constrained to be at least the harvest volume in the previous period.

The EDFs for wildlife habitat developed in Section 4.3 are used to set constraint levels on habitat levels for each of the wildlife species. Runs are made where the habitat area for all species are simultaneously constrained to be at least a specified quantile of the probability distribution of the 5+ habitat in each of the periods of the planning horizon. The quantile constraints used are 0.0, 0.025, 0.05, 0.10, 0.125, 0.15, 0.20, 0.25, 0.30, 0.35, 0.40, 0.45, and 0.50. The 0.0 quantile represents no habitat constraints (pure timber emphasis). All the optimization runs for habitat quantile constraints of 0.45 or greater were infeasible. This indicates that it is impossible to simultaneously provide habitat areas for all five species at the 0.45 quantile level: there are trade-offs between species.

The system was modeled using the Woodstock forest modeling package (Remsoft Inc., 1998) and solved using the C-WHIZ linear programming (LP) solution software (Ketron Management Science, 1998). Woodstock provides a convenient way of specifying a forest management problem using a flexible syntax. It can generate a LP matrix as input to solution software such as C-WHIZ and translate the LP solution into easily understandable summary tables and graphs. A five-year period, and a 25 period (125 year) planning horizon was used for all the Woodstock models. The extra five periods (25 years) relative to the simulations were added to the planning horizon in order to minimize the effect of end of planning horizon timber anomalies typical of optimization-based timber harvest scheduling models without ending inventory constraints.

Habitat projections under management, relative to the projected 95% confidence limits for natural conditions, are summarized in Figure 4.9 for habitat quantile constraints of 0, 0.025, and 0.25. In the timber emphasis run (quantile constraint level 0) good (5+) PIMA habitat is eliminated fairly quickly. This occurs because ideal PIMA habitat occurs in older white spruce stands. Because of the value and volume of timber in these stands, they are also prime candidates for logging. The areas of good habitat for BWHK, TTWP, and BTGW also fall below the lower confidence limit for natural conditions when the forest is managed without consideration of habitat. Habitat for the MEVO under all quantile constraint levels is well within the confidence intervals for natural conditions.

The optimization problem is infeasible for all habitat constraint levels of 0.45 or greater with and without non-declining yield constraints. This means that it is not possible for the forest to simultaneously provide good habitat for all five vertebrate species over all periods in the planning horizon. The habitat area for any species can only be increased at the expense of another species.

The trade-offs between timber supply and habitat considerations are summarized in Figures 4.10 – 4.14. These graphs also show the effect of the relaxation of non-declining yield constraints. With non-declining timber yield constraints in place, the net present value of forest management activities is \$ $1.363 \times 10^9$ with no habitat quantile constraints. The NPV declines to \$ $0.621 \times 10^9$ at the 0.40 habitat constraint level, less than half the unconstrained value. Hardwood and softwood harvest volumes also decline with the imposition of tighter habitat area constraints.

A useful output of most LP solvers is a listing of the shadow prices of each of the constraints in the model. The shadow price of a constraint represents the increase in the objective function value that could be achieved if the constraint was relaxed by one unit, all other things being equal. Figures 4.15, 4.16, and 4.17 present the shadow prices by period for habitat constraints set to the quantiles 0.025, 0.250, and 0.400. In all cases the shadow prices are expressed on a per hectare basis. And, in all cases, the shadow price for MEVO habitat is zero.

For the quantile 0.025 constraint (Figure 4.15), the largest shadow prices are associated with PIMA and BTGW in the early periods of the planning horizon. This is not surprising as ideal PIMA habitat is overmature white spruce and ideal BTGW habitat is overmature mixed stands. The yield curves in Figures 4.1 and 4.2 show that total volume declines with increasing age for overmature timber. Net present value maximizers would prefer not to let this timber fall down and rot, so there is an incentive to log and regenerate these stands. Not harvesting the stands in order to provide PIMA and BTGW habitat therefore has a cost.

The shadow prices increase dramatically in the earlier periods when the habitats are constrained to the 0.250 quantile (Figure 4.16). The constraints on PIMA and BTGW are still costly, but constraints on BWHK habitat are extremely costly in periods 1 and 3. BWHK prefers overmature aspen and mature mixed stands. These stands are also prime

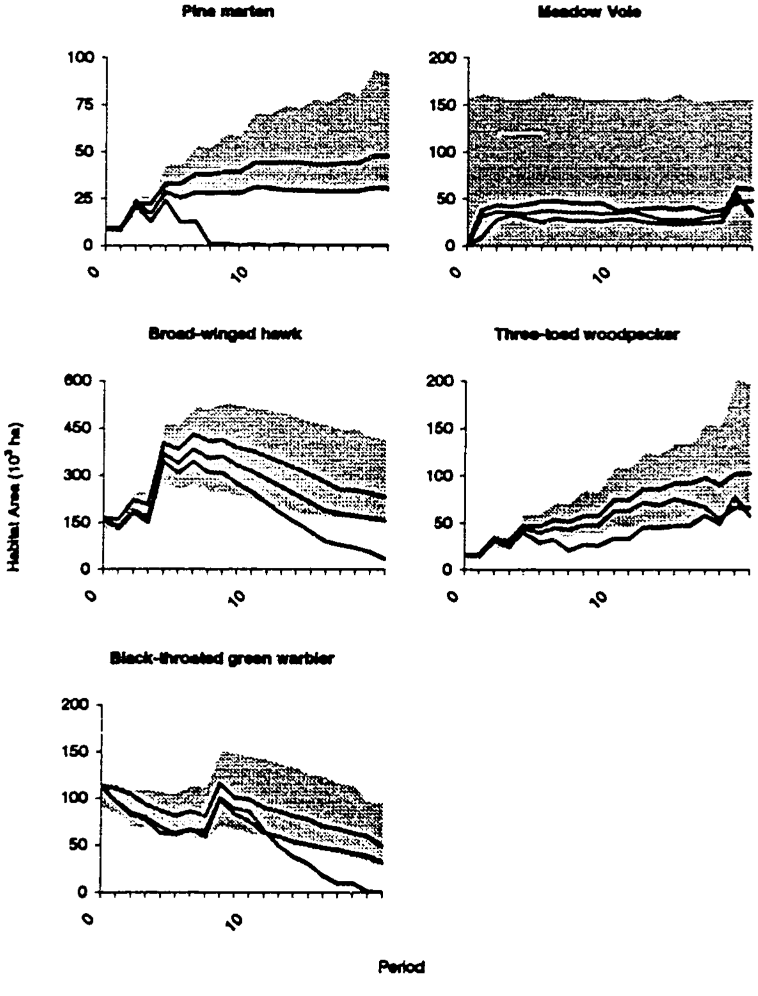


Figure 4.9: Habitat projections under habitat quantile constraints of 0, 0.025, and 0.25. The 95% confidence limits are shaded grey.

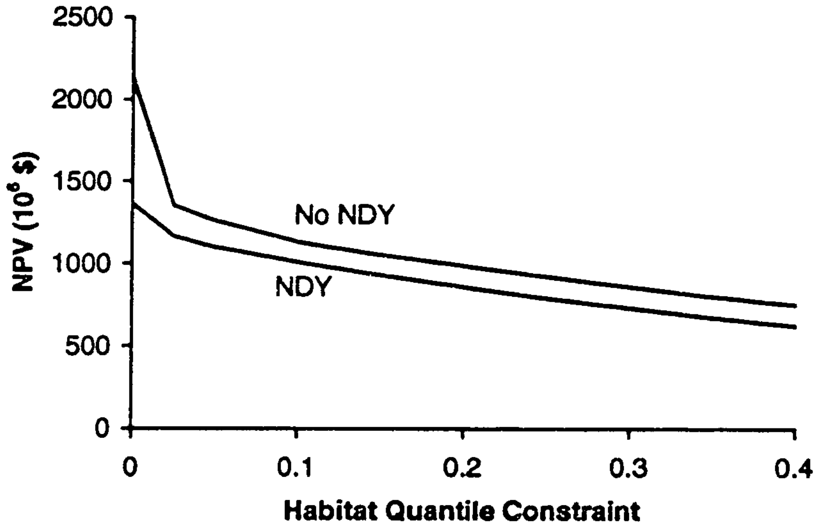


Figure 4.10: Net present value by quantile constraint level for runs with and without non-declining yield constraints.

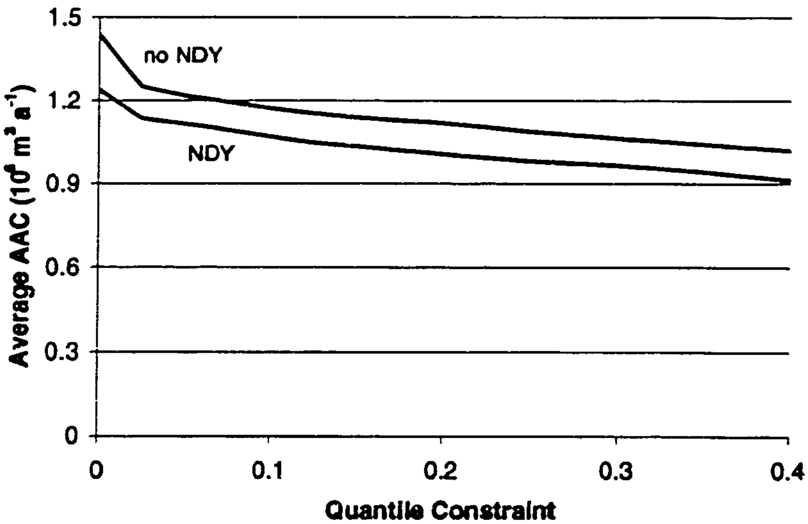


Figure 4.11: Average hardwood annual allowable cut by quantile constraint level for runs with and without non-declining yield constraints.

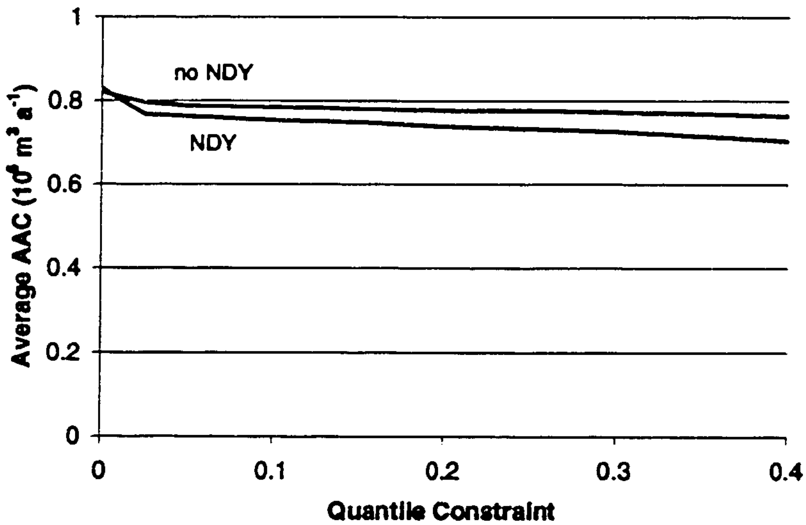


Figure 4.12: Average softwood annual allowable cut by quantile constraint level for runs with and without non-declining yield constraints.

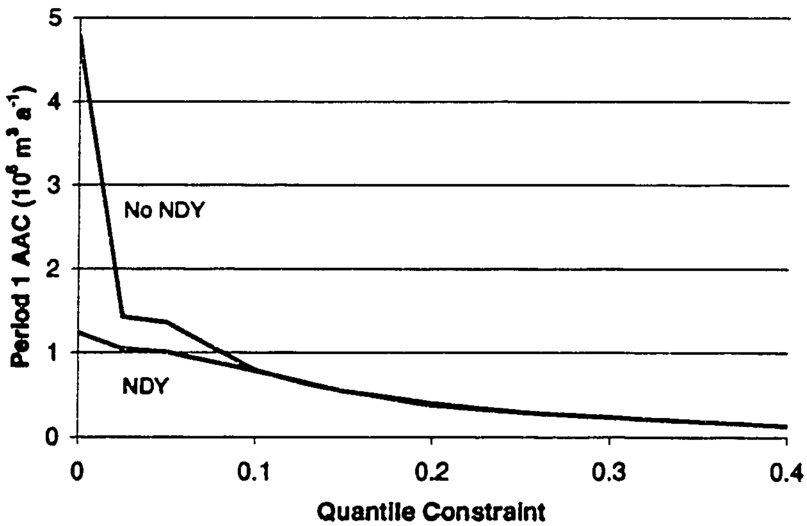


Figure 4.13: First period hardwood annual allowable cut by quantile constraint level for runs with and without non-declining yield constraints.

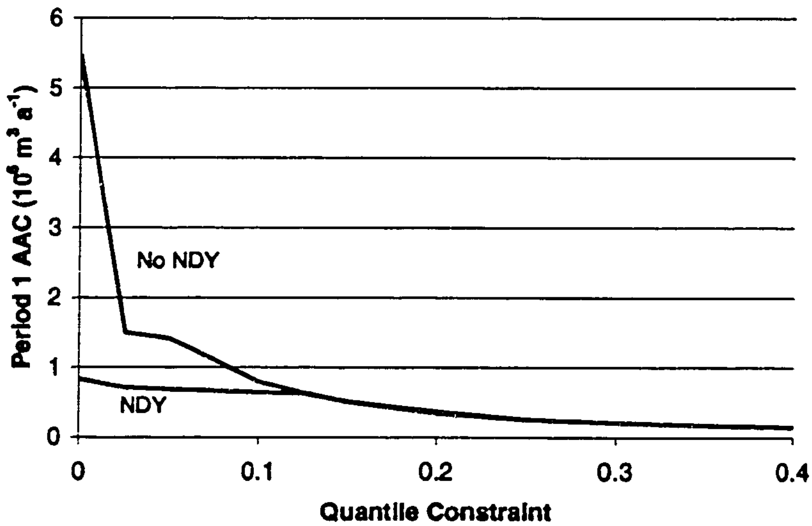


Figure 4.14: First period softwood annual allowable cut by quantile constraint level for runs with and without non-declining yield constraints.

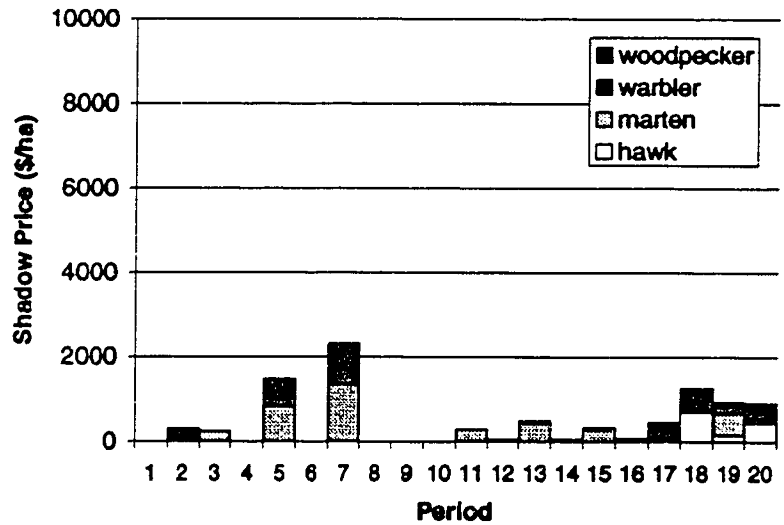


Figure 4.15: Shadow prices by 5-year period for the 0.025 quantile constraint run. The shadow price for MEVC habitat is zero in all periods.

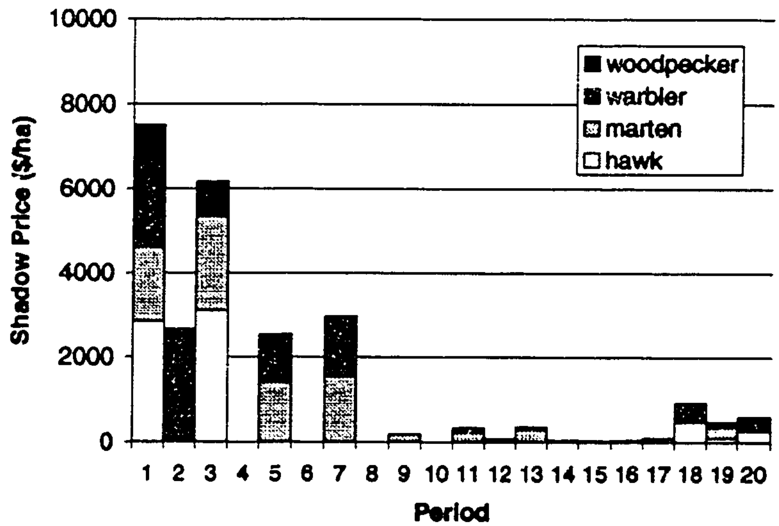


Figure 4.16: Shadow prices by 5-year period for the 0.25 quantile constraint run. The shadow price for MEVO habitat is zero in all periods.

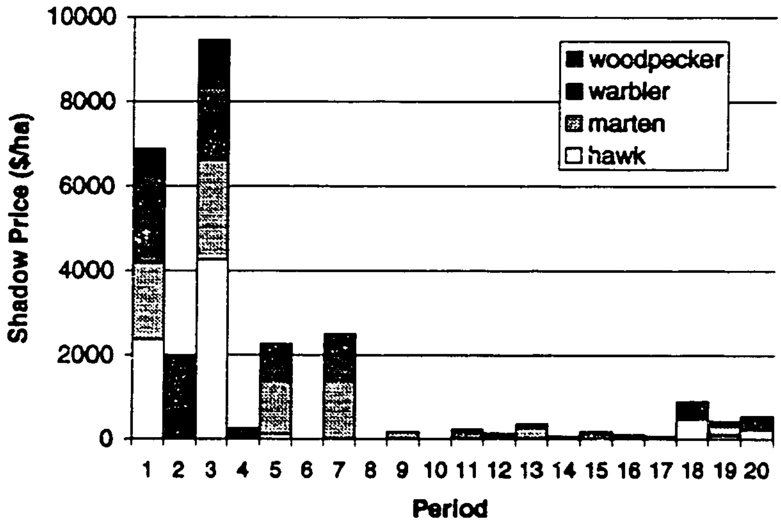


Figure 4.17: Shadow prices by 5-year period for the 0.40 quantile constraint run. The shadow price for MEVO habitat is zero in all periods.

candidates for logging.

Essentially the same story can be told for the quantile 0.40 constraints (Figure 4.17) except that the BWHK constraints in period 3 are noticeably more expensive.

The shadow prices provide important information on the costs of constraints. They may help identify areas for considering alternative management strategies. For example, if BWHK habitat is particularly costly in a period, it may be possible to enhance BWHK habitat through means other than maintaining a particular age class of forest.

## 4.. Concluding Comments

This chapter presented a set of models that could be used to help forest managers determine the appropriate harvest level for a forest managed under the natural disturbance model. I argue that the natural rate of disturbance and equilibrium age class structures are inappropriate characterizations of the natural disturbance regime of the boreal mixedwood and should not be used.

As an alternative, I use Monte Carlo simulation to project the variability in habitat for five species of vertebrates for forest subject to a lognormal natural fire regime. The quantiles from these projected distributions are used to set constraint levels for a linear-programming based optimization model. This model is used to develop a curve representing the trade-off between financial objectives and habitat quantiles. One of the most important decisions that the forest manager will have to make is which point on the trade-off curve represents the appropriate goal for management.

The main advantage of the modeling system presented in this chapter is that it explicitly recognizes the variability in a highly variable ecosystem. The system is used to help identify the trade-offs between competing goals in the context of natural disturbance management.

There is substantial room for improvement and development of the modeling system used here. The optimization runs used here were all deterministic. Because it is unlikely that fire suppression efforts will ever eliminate the risk of forest fire, it would be useful to incorporate a stochastic optimization procedure into the system. Both the simulation and optimization components of this study were aspatial. Wildfire behavior and wildlife habitat certainly have spatial components. Addition of some level of spatial detail to the modeling system may be worth considering. I believe however, that many of the important trade-offs can be captured using an aspatial modeling system such as the one presented here.

In the models used here, each of the vertebrate species used as indicators is given equal weight in the sense that habitat quantile constraints are applied to all species at the same level in all runs. This is consistent with the ideas behind the coarse filter approach to ecosystem management in that no species is assigned a greater weight than any other. However, this system could potentially be used in a public participation context to develop alternatives to help elicit the preferences of the public for different alternative future forests. The alternative future forests could be described in terms of stocks and flows of financial values, recreation opportunities, abundance of habitat for different wildlife species, and other forest values.

# Bibliography


Armstrong G.W. 1999. A stochastic characterisation of the natural disturbance regime of the boreal mixedwood forest with implications for sustainable forest management. Can. J. For. Res. **29**: 424–433.

Cumming S.G., Burton P., Prahacs S., and Garland M. 1994. Potential conflicts between timber supply and habitat protection in the boreal mixedwood of Alberta, Canada: A simulation study. Forest Ecology and Management **68**: 281–302.

Davis L.S. and Johnson K.N. 1987. Forest Management. McGraw-Hill, Inc., New York, 3rd edn.

Hof J. and Bevers M. 1998. Spatial Optimization for Managed Ecosystems. Complexity in Ecological Systems. Columbia University Press, New York.

Ketron Management Science. 1998. C-Whiz: Linear Programming Optimizer. Version 3. Ketron Management Science, Arlington, VA.

McNichol M.K., Stepney P.H.R., Boxall P.C., and Spaldine D.A.E. 1981. A bibliography of Alberta ornithology. Tech. rep., Provincial Museum of Alberta.

Remsoft Inc. 1998. Woodstock User's Guide. Remsoft Inc., Fredericton NB, version 1.0 edn.

Van Wagner C.E. 1978. Age-class distribution and the forest fire cycle. Can. J. For. Res. **8**: 220–227.

# Chapter 5

# Discussion and Conclusions

This thesis presented three studies related to the natural disturbance model of forest management (Hunter, 1993). The first study (Chapter 2, and a version published as Armstrong *et al.* (1999)) projected the consequences on timber supply of a simple interpretation of the natural disturbance model: the rate of disturbance (harvest and wildfire) under management should be set equal to the rate of natural disturbance, and the proportion of different forest types subject to managed disturbance should be equal to that under natural disturbance. The study compared the consequences of three very different estimates of the natural disturbance rate (0.5, 1.0, and 2.0%/year) to the timber supply that would result from implementation of the current sustained yield policy. Implementing the natural disturbance model would result in major reductions in annual allowable cuts.

The second study presented in this thesis (Chapter 3 and a version published as Armstrong (1999)) began as an attempt to reconcile the differences in the natural disturbance rates estimated by Cumming (1997) and Murphy (1985) for the boreal mixedwood forest of northern Alberta. The magnitude of the differences is important because of the economic and ecological consequences of the harvest rate decision. If the rate of harvest is based on the rate of natural disturbance, the estimate of the natural disturbance rate is critical information. Chapter 3 presented a characterization of the natural disturbance regime for the study area. The annual area burned was well-described as a serially independent random draw from a lognormal distribution. Monte Carlo simulations were used to identify some of the consequences of this natural disturbance regime. Estimates of the mean annual burn rate for the study area are highly variable. This makes single parameter characterizations of the disturbance regime (*e.g.* mean disturbance rate, fire cycle) unreliable. There is no equilibrium age class structure for a forest subject to the lognormal disturbance regime parameterized in Chapter 3. This means that a particular target age class distribution cannot be justified on the basis that it is the "correct" natural distribution.

The results of Chapter 3 suggested that average disturbance rates and target age class distributions are inappropriate guides for harvest rate determination under the natural disturbance model for the boreal forest because of the extreme interannual variation in the natural disturbance rate. Chapter 4 presented a modeling system that recognized the variability inherent in the disturbance regime, and used that variability to help guide harvest rate determination. The forest was described in terms of habitat area for selected vertebrate species. The forest was subjected to the natural disturbance regime presented in Chapter 3 for a 100-year planning horizon. This process was repeated 1000 times to generate a projection of the probability distributions of habitat areas for each year of the planning horizon. These probability distributions were used to set constraints for an optimization based forest planning model, which was used to develop trade-off curves between objective function values and constraint levels. The trade-off curves could be used by the owners and managers of the forest to decide what level of trade-offs are acceptable.

This thesis has contributed to the understanding of the natural disturbance regime of the boreal mixedwood forest, and has suggested a way that knowledge of the regime could be used to help a forest manager determine the appropriate rate of harvest.

There is substantial room for further research. The disturbance model presented in Chapter 3 is aspatial and independent of forest composition. The model of the disturbance regime could be refined to incorporate information on the composition and spatial arrangement of stands in the forest. This would allow for evaluations of the effectiveness of different fire control strategies, for example.

The optimization model used in Chapter 4 is a deterministic model. It would be worthwhile exploring stochastic optimization techniques for application to this problem. Chapter 4 presents the forest planning problem as a choice between different combinations of financial outcomes and levels of wildlife habitat and timber production. The techniques used in that chapter could be used to develop trade-off curves to help elicit from the public what combination of goods and services should be provided from the forest. This public involvement process could help in the specification of the "desired future forest" advocated by the Alberta Forest Management Science Council (Alberta Forest Management Science Council, 1997).

# Bibliography


Alberta Forest Management Science Council. 1997. Sustainable forest management and its major elements: Advice to the Land and Forest Service on timber supply protocols.

Armstrong G.W. 1999. A stochastic characterisation of the natural disturbance regime of the boreal mixedwood forest with implications for sustainable forest management. Can. J. For. Res. **29**: 424–433.

Armstrong G.W., Cumming S.G., and Adamowicz W.L. 1999. Timber supply implications of natural disturbance management. For. Chron. **75**: 497–504.

Cumming S.G. 1997. Landscape dynamics of the boreal mixedwood forest. Ph.D. dissertation, Department of Forest Sciences. University of British Columbia, Vancouver.

Hunter M.L. 1993. Natural fire regimes as spatial models for managing boreal forests. Biol. Conserv. **65**: 115–120.

Murphy P.J. 1985. Methods for evaluating the effects of forest fire management in Alberta. Ph.D. dissertation, Department of Forestry. University of British Columbia, Vancouver.