Combined forest management effect on landscape carbon stock changes in west-central Canada

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Executive Summary

INTRODUCTION AND SCOPE OF INVESTIGATION

Increased human activities have changed global ecosystems including the dynamics of C stocks in forestlands. Climates across the world have changed in an acceleratory manner, and the CO_2 concentration in the atmosphere has already increased by about 30% since the industrial revolution started. The international community has been intensively working on a consensus of how the environmental conditions could be managed for a healthy and sustainable development, and a central issue is to understand how different human activities might influence ecosystem dynamics such as changes in C stocks in forest lands.

Dynamics of C stocks in forest lands are determined by the sizes of living biomass, dead woody materials, and soil C pools. A relatively sound understanding has been achieved in the dynamics of living biomass C pools, and thus it is used in some of the international reporting. In this report, we shall focus on C dynamics in living biomass in forest lands and explore how it could be influenced by different management operations.

The normal forest growth and decline processes will be influenced by various natural and anthropogenic disturbance regimes. In the boreal forests of Canada, major natural disturbances include fire, insect, disease, wind, and climate-induced mortality. We focus on fire and harvest regimes in this study and will address mountain pine beetle influences elsewhere.

The objective of our current research is to provide a case study on how the long-term C stock dynamics at the landscape scale could be influenced by different combinations of forest management options.

METHODOLOGY

Owing to the limited availability of empirical observations, our case study uses a modeling investigation approach that has been the most often used method in climate change and C dynamics research. We provide an overview of existing modeling approaches, which include spatially implicit, spatially distributed, and spatially explicit models. We used the spatially explicit modeling approach in our study.

Based on the forest conditions of Fort A La Corne (FALC) in central Saskatchewan, this case study employed one scenario and one process-based model to simulate C stock dynamics under various combinations of forest fire and harvest regimes. The Woodstock software package, which is a commonly used tool for operational timber supply analysis, is used as the scenario model to simulate the forest and C dynamics influenced by both harvest and fire regimes. The process-based spatial fire regime model SEM-LAND is used to simulate forest and C dynamics under natural conditions (without human interventions), which represent the conditions under the forest policy of emulating natural disturbance patterns in harvest planning.

Simulated forest and C dynamics under various forest management options are then compared with the estimates using default and average values used in international reporting, which were approved by the Intergovernmental Panel on Climate Change (IPCC), to identify the conditions where forest productivity and C sequestration could be enhanced.

SUMMARIZED RESULTS

Changes in the C sink size are largely attributed to the dynamics of forest age distribution. Our simulation results indicate that the mean forest age can be increased gradually until 180 years old at the simulated 120-years and then decline; the mean forest C stock per hectare can be increased in the first 50 years and then gradually decrease for the following 100 years, and after that the C stock will increase again to recover the initial C stock level at the last planning period. The different dynamic patterns in mean forest age and mean C stock are caused because the mean annual increment (MAI) of volume is not constant over time. Our results suggest that forest productivity and C sink size cannot be sustained with complete protection against all disturbances.

Under different harvest rates expressed as annual allowable cut (AAC), similar patterns of mean forest age change over time, but the highest mean forest age decreases with increasing AAC. When examining the volume change, a decreased value can be expected when the AAC becomes larger. The mean values over 20 planning periods and 10 replications show the trends of forest and C stock dynamics under various AACs. The trends were summarized in mean forest age, total forest inventory, volume per hectare, and old forest area; and they all decrease with increasing AAC.

Under different fire regimes expressed as the fire cycle, the total mean inventory, mean forest age, and old forest area (across 20 planning periods and 10 replications) can rise with increased length of fire cycle, because the mean burned area decreased with increasing fire cycle length.

With a systematic model experiment that simulates forest dynamics under different combinations of fire and harvest regimes, we made the following observations:

- Not all of the combinations would generate meaningful results because some simulations would be infeasible before completing the 20 planning periods. High AAC and short fire cycles would tend to contribute to unsuccessful simulations.

- The mean forest age would be the highest with no harvest and the longest fire cycle, and it would drop with decreasing fire cycle and increasing harvest rates. The mean C stock per hectare would be positively correlated to mean forest age.

- MAI and mean annual C sequestration would both increase with greater disturbance rates, and this would be true for both harvest and fire regimes.

- The mean C sink size (subtracting C released by fire from C sequestration) under various combinations of fire cycles and AACs would represent a mix of positive and negative values. Short fire cycles would tend to make the C sink size negative thus representing a C source.

- In most cases the FALC forests would function as C sinks except when the fire cycle becomes very short (< 100 years). The forests would tend to function even more as a C sink if the C transferred from living biomass to the forest products pool were considered in the C accounting process. Therefore, the area has the potential to positively contribute to the C sink.

CONCLUSIONS AND RECOMMENDATIONS

We can link our simulation results to the role of forests in the global C budget and explain the usefulness of current research results in international reporting. The following points summarize the lessons learned from our modeling investigation:

- Complete protection from any disturbance might not be the best strategy for an enhanced C stock in forest lands, and existence of disturbances might not necessarily be bad for C dynamics.

- Forest age distribution and its dynamics are important in determining forest growth rates such as MAI, and thus the C sequestration, in which a certain range of mean forest ages would result in values higher than the default and average values recommended by the IPCC. Therefore, keeping forests in the age classes with a high MAI could serve as a target for forest managers.

- Furthermore, emulating natural fire patterns in harvest planning could increase MAI in young and mid-aged forests, and the C sequestration could increase significantly from 0.46 t/ha, the default value for boreal forests, to up to 1.2 t/ha.

- If the C transferred from living biomass to the forest products pool is considered in the C accounting process, the FALC forests will tend to function even more as a C sink.

We recommend future research in the following areas:

- Carry out more case studies using operational forest inventory data and involve the wood/timber supply research community for international reporting purposes.

- Incorporate mountain pine beetle regime effect within a unified simulation framework.

- Perform a comprehensive economic analysis before implementing research results in forest management practices.

- Include C dynamics in dead woody materials and soils that could enable forest managers and researchers to address the question of whether the C released through decomposition and respiration would be large enough to offset C sequestration in living biomass, with the intent that the role of forests in global C budget could be redefined.

Abstract

Increased human activities have changed global ecosystems including the dynamics of C stocks in forest lands, which are determined by the sizes of living biomass, dead woody materials, and soil C pools. The current study focuses on C dynamics in living biomass in which a relatively consistent understanding has been achieved and has been used in international reporting. Based on the forest conditions of Fort A La Corne (FALC) in central Saskatchewan, this case study employed one scenario and one process-based model to simulate C stock dynamics under various combinations of forest fire and harvest regimes. Our simulation results suggest that forest productivity and C sink size cannot be sustained with complete protection against all disturbances. Changes in the C sink size are largely attributed to the dynamics of forest age distribution. Forest management options that keep forests within a certain range of mean forest age could result in both higher mean annual increment (MAI) and C sequestration rates than the default and average values used by Intergovernmental Panel on Climate Change. Implementation of emulating natural fire patterns in harvest planning can increase MAI in young and mid-aged forests, and the C sequestration can thus increase significantly. Our results suggest that in most cases the FALC forests will function as C sinks except when the fire cycle becomes very short (< 100 years); hence, the area has the potential to positively contribute to the C sink. If the C transferred from living biomass to the forest products pool is considered in the C accounting process, the FALC forests will tend to function even more as a C sink. We recommend future research include more case studies using operational forest inventory data and involving the wood/timber supply research community for international reporting purposes; incorporate mountain pine beetle regime effect within a unified simulation framework, and perform a comprehensive economic analysis before implementing research results in forest management practices.

1. Introduction

Forests absorb CO_2 from the atmosphere through photosynthesis during their growth process. The world's forests store about 60 gigatons (billion tons) of carbon (C) from the atmosphere every year (Brown 1996). Increased human activities have changed global ecosystems including climate change and the dynamics of C stocks in forest lands. Research results have shown that climates across the world have changed in an acceleratory manner, and the CO_2 concentration in the atmosphere has already increased by about 30% since the industrial revolution started; this change is largely attributed to the increased use of fossil fuels (coal, oil, and natural gas) for energy (IPCC 2001). These observations have been supported and explained by a number of general circulation models and global climate models (GCMs). Despite the uncertainties involved, the international community has been intensively working on a consensus of how the environmental conditions could be managed for a healthy and sustainable development. The effort can be traced back from the pioneer document "Our common future" (Bruntland et al. 1987), to the international treaty of United Nations Framework Convention on Climate Change (UNFCCC) in 1992, to the Kyoto Protocol (KP) that sets goals of greenhouse gas (GHG) emission for the industrialized countries in 1997, and to a parallel effort by the Asia-Pacific partnership on climate change in 2005. In all of these efforts, a central issue is to understand how different human activities might influence ecosystem dynamics such as changes in C stocks in forest lands.

Dynamics of C stocks in forest lands are determined by the sizes of living biomass, dead woody materials, and soils C pools (Penman et al. 2003). Among these C pools, research results reported in the literature have shown that relatively consistent understanding has been achieved in the dynamics of living biomass C pool, and the understanding of the C dynamics in woody materials and soils has remained inconsistent. Consequently, only the dynamics of C in living biomass is required in some of the international reporting such as in KP (Penman et al. 2003). In this report, we shall focus on the C dynamics in living biomass in forest lands and explore how it could be influenced by different management operations.

Conceptually, normal forest growth and decline processes will be influenced by various natural and anthropogenic disturbance regimes. In the boreal forests of Canada, major natural disturbances include fire, insect, disease, wind, and climate-induced mortality. In western Canada, fire remains the most destructive natural disturbance on forests. Timber harvest is the major anthropogenic disturbance regime that has a direct influence on forest dynamics, and the management of natural disturbance regimes has an indirect influence on forest dynamics. Major insect pest species in this region are mountain pine beetle (MPB), spruce budworm, and forest tent caterpillar. Among these species, MPB has the most devastating effect on forest dynamics and has been in an outbreak phase for the last several years (BC Ministry of Forests 2003). The MPB attacks most pine species in BC, especially the lodgepole pine. With the climate warming in western Canada, this species has rapidly expanded its infestation area into northern BC and western AB where lodgepole pine is distributed (Alberta Sustainable Resource Development 2004; Li and Barclay 2004; Li et al. 2005). Furthermore, there is a serious concern on whether MPB would also attack jack pine under environmental pressure, because if this happens the MPB infestation could quickly spread to the rest of Canada and the damage to the wood supply and C dynamics might be significant (Les Syfranik, Pacific Forestry Centre, Canadian Forest Service, personal communication 2004). In this report, however, we shall focus on fire and harvest regimes, and address the MPB influence elsewhere.

National scale reporting for the UNFCCC and KP essentially require aggregating results from analyses carried out at smaller (regional and provincial) spatial scales. Aggregation of the results can be achieved through scaling-up stand-level understanding and dynamics to landscape and provincial scales. In this scaling-up process, major challenges can be attributed to the landscape scale where most of the spatial processes that influence forest dynamics are operating. Contemporary ecological theory has revealed that the dynamics of whole ecosystems might not be simply equal to the sum of their parts due to the non-linear relationships embodied in the hierarchically structured complex ecosystems (Costanza and Jørgensen 2002). Therefore, the methods of scaling estimated C stock dynamics at smaller spatial scales up to larger spatial scales might be one of the challenges forest managers and researchers are facing. Furthermore, reliable assessment of living biomass C stock dynamics at smaller spatial scales could only be obtained from accurate estimates of merchantable forest volume dynamics because of the close relationships between merchantable forest volume and aboveground biomass, and between above- and below-ground biomass estimates (von Mirbach 2000). Due to the importance of methodological issues in C dynamics research, we shall present an overview on the topic and describe the approach that is being used in the current study in Section 2 to ensure our approach is consistent with modern ecological understanding.

The objective of our current research is to provide a case study on how the long-term C stock dynamics at the landscape scale could be influenced by different combinations of forest management options. In Section 3, we shall describe the study area of the Fort A La Corne (FALC) in central Saskatchewan. The two models and the model experiments as well as the analysis methods will also be summarized in this Section. We shall present our results in Section 4, followed by a discussion Section 5, and a conclusion Section 6 with recommendations for future research.

2. Overview of Simulation Modeling Approaches

Owing to the limited availability of empirical observations, the simulation-modeling approach has been the most often used method in climate change and C dynamics research. Modeling approaches, however, can be vary tremendously because of the questions and issues in mind at the time of model development. Consequently, not all models can address the same issue properly, and model users need to be aware of the assumptions made and model strengths and weaknesses before drawing any conclusions. In this section, we provide an overview of existing modeling approaches, followed by a description of the approach used in the current research.

2.1. Modeling approaches in C dynamics research

Simulation models are essentially assemblies of the model designers' understanding on how the system works. The models can be theoretical or operational depending on the type of information databases are being used in the models. Simulation models for theory development and exploration usually apply assumptions based on conceptual understanding and these assumptions may or may not be necessary to achieve a solid test. In operational simulation models, however, the assumptions should be tested before being used to avoid misleading researchers in addressing policy and practical forest management issues. This is because the model behavior or simulation results are simply representations of the logical outcomes of these assumptions.

Input data structure is one of the main constrains in the selection of a simulation modeling approach. For example, spatial modeling approaches would be very difficult, if not impossible, to implement for non-spatial categorical data. Input data structure can also impose constraints on methods of data processing and result analysis. Statistics from spatial analysis such as spatial pattern indices can only be obtained from spatial data but cannot be estimated from non-spatial data.

Simulation models using spatial data as input can process the data with different methods. Spatially implicit, spatially distributed and spatially explicit are the three main simulation approaches in this category. With the belief that the forest dynamics in sub-spatial units of a large forest landscape are completely independent of each other, a spatially implicit modeling approach simulates forest dynamics in these sub-spatial units separately and sums them together to obtain the total forest landscape dynamics. This approach is apparently suitable for forest dynamics that are not significantly influenced by spatial processes. Examples are stand-level forest growth without suffering large and irregular fire and insect disturbances. This approach has been widely applied in ecosystem dynamics such as the individual-based modeling approach (Grimm and Railsback 2005; Keane et al. 1995, 1996; Keane et al. 1996). The main advantages of employing this modeling approach are their straightforward nature and maximized computing power usage such as parallel computing technology. In some cases, this modeling approach has been even simplified to simulating a few representatives of typical forest types and then adding them up according to the respect areas in these types of forests. The assumption behind this simplification was that the statistical properties of each type of forests could be obtained from the average of a certain number of samples (i.e., multiple simulations). This assumption holds in statistical sense, as long as the simulated forest stands are suitable representatives. In this case, forest dynamics in a large region can be estimated from simulations performed in much reduced sub-spatial units. Consequently, the time required to obtain estimation of forest dynamics could be greatly reduced.

The faith behind the spatially distributed simulation modeling approach is the same as the spatially implicit approach. However, this modeling approach tried to speed up the simulations through using look-up-table and response surface technology. Examples of this modeling approach includes the 3-PG model (Landsberg and Waring 1997).

The spatially explicit simulation modeling approach does not exclude the possibility of significant influence of spatial processes on forest landscape dynamics. Examples of the spatial process include fire spread, and insect and seed dispersal. Simulations of the spatial processes are implemented in a spatially explicit way that requires no or fewer assumptions on the processes. For example, simulated exact locations, sizes and shapes of individual fire events can be shown on forest landscape maps and thus their impact on forest dynamics could be evaluated with less uncertainty. Therefore, as long as the statistical properties of a fire regime (i.e., all fire activities within a given region over a period of time) are consistent with modern conceptions, the simulation results with this modeling approach will have higher capability of capturing scenarios in the real world. The disadvantage of employing this modeling approach is the requirement of a large computing power for completing the simulations within a reasonable timeframe. This large computing power is due to the detailed simulation of spatial propagation or dispersal processes. Furthermore, parallel computing technology may not be able to help in this case, due to the large information exchange requirement at the boundaries among different small regions (Wu et al. 1996).

Despite the technical differences, the important question is whether employing different modeling approaches could significantly influence the estimated forest dynamics. Very few studies have been reported in the literature to address this question. Nevertheless, an example has been documented by

Li and Apps (1995, 1996). In an effort to adapt the FORSKA gap model, which was originally developed for Scandinavian boreal forests, to Canadian forest conditions, researchers have tried to explain why the simulation results were not consistent with observations in the Canadian boreal forests. Two possible reasons were suspected: inappropriate parameter values for Canadian tree species or unsuitable representation for disturbance effect. At the time, the best information on Canadian tree species had been incorporated, so the research focus was on disturbance effect. A model experiment was conducted to examine the effect of different fire modeling approaches on simulated forest biomass dynamics. The simulation results indicated that spatially implicit simulation approach could result in the biomass burn significantly higher than that of spatially explicit modeling approach (Li and Apps 1996). The reason is that when fire probability is related to forest age or time-since-last-burn, older forests will always have a higher probability to be burned than younger forests in the spatially implicit modeling approach, and more biomass would be burned in an older forest stand than in a younger stand. Consequently, while the statistical results on simulated fire regimes remained the same, their impact on biomass dynamics showed a significant difference. With the spatially explicit modeling approach, the simulated forest stands neighboring the disturbance center will have a higher probability of burning, regardless of their current age. Consequently, slope of the original age-dependent fire probability function will be reduced under the spatially explicit modeling approach. The results reported in Li and Apps (1996) suggested, “the explicit inclusion of spatial interactions is required in order to properly represent the relationships between spatial pattern and temporal dynamics in the boreal forests of Canada”. So, if the spatial fire process is simplified as non-spatial, one could expect an under-estimate in the forest biomass dynamics. With these earlier results in mind, our research on fire regime simulations and forest landscape dynamics, a spatially explicit modeling approach is employed.

In the spatially explicit modeling approach, forest landscapes are usually characterized as an array of rectangular cells or pixels with each representing a homogeneous stand in cover type, age, and density class. Proper methodologies of fire simulations are important in forest landscape dynamics subject to large and irregular fire disturbances such as in Canada’s boreal forests. The section 2.2 provides an overview of spatial fire regime simulations.

2.2. Spatial fire regime simulations

A fire regime characterizes the fire activity or fire pattern for a given region (Merrill and Alexander 1987), resulting from long-term interactions among fire events, landscape structure, fuel and weather conditions, and human activities (Turner et al. 1994; Li 2000a, 2000b). It can be described by a number of descriptors (Weber and Flannigan 1997), and Li (2002) categorized these descriptors into two groups. The first group includes the descriptors related to fire occurrence and frequency such as fire number, frequency, cycle, size, and season, and the second group consists of descriptors related to fire effect such as fire type, intensity, and severity. Each of these descriptors might reflect one aspect of a fire regime as a whole, and hence they might be potentially interrelated with each other. For example, it has been demonstrated that the fire frequency and size distribution might be correlated with each other under natural conditions (Li et al. 1999). The natural conditions refer to the situations where fires were allowed to spread freely without human intervention. Despite the use of diversely defined fire frequency and fire cycle in fire sciences and forest dynamics, they could be also interrelated from a computational perspective (Li 2002). It has also been documented that the dynamics of forest age structure were also related to the fire disturbance patterns (Van Wagner 1978; Li and Barclay 2001). Due to the self-organization property of ecosystems (Holling et al. 1996), landscapes (Bradbury et al. 2000), and fire regimes (Malamud et al. 1998), different temporal fire ignition source patterns displayed in historical fire records might not lead to significant differences in fire cycles under natural conditions (Li 2000c). However, the understanding of spatial and temporal dynamics of fire regimes is still incomplete; this is especially evident when considering how different fire suppression options could influence the dynamics of a fire regime and their associated forest ecosystems.

A fire process can be simulated in two stages: fire initiation and spread (Li and Apps 1995, 1996). The fire initiation stage starts from the presence of a fire ignition source in a given cell until the fire kills most trees in the cell; this is called a fire initiation. Once a fire is initiated, the fire would have the potential to spread, thus the fire spread stage begins. The fire would spread continuously until it stopped because of non-flammable or low flammability cells, or because it reached the boundary of the study area, and thus the fire spread stage was completed.

2.2.1. Fire Initiation Simulation

At least three issues need to be considered in the simulation of fire initiation: annual number of fire ignitions, spatial locations of the fire ignitions, and the relationship between the probabilities of fire initiation and spread.

Historical fire records show that the annual number of fire ignitions for a given region was not static but dynamic, fire cause-dependent, and region-dependent (Canadian Council of Forest Ministers 1997; Li 2000c). In the temporal dynamics of fire ignition sources, 3- to 4-year cyclic patterns were reported in the literature, e.g., Granström (1993) for Sweden, Cumming et al. (1995) and Li (2000c) for Alberta, and Li et al. (1997) for Northwestern Ontario. However, the mechanisms for such a cyclic pattern still remained unclear.

In spatial fire regime simulations, various assumptions were made regarding the annual number of fire ignitions including a fixed number of fires per year (e.g., Holling et al. 1996), correlated to climate conditions (e.g., Gardner et al. 1996), random (represented by a normally distributed random variable) fire numbers around a mean value (e.g., Li 2000a, 2000c), and cyclic patterns (e.g., Li et al. 1997; Li 2000c). Using a process-based fire regime model Spatially Explicit Model for LANdscape Dynamics (SEM-LAND), a comparison of simulation results between random fire numbers and cyclic patterns of fire ignition sources indicated that there was no significant difference in simulated fire cycles when the level of annual fire ignition source is high (0.6 fire/year/100 km^2) (Li 2000c). Nevertheless, significant differences could be expected when fire suppression is involved and the level of annual fire ignition source is low (0.3 fire/year/100 km^2).

The implementation of annual fire ignition sources in spatial fire simulations has also been diverse, ranging from fixed ignition locations (e.g., Peterson 1994; Hargrove et al. 2000) to selections from historical ignition locations (e.g., Gardner et al. 1996) to randomly determined locations (e.g., Li 2000a, 2000c). These assumptions were made for different simulation purposes, and could be regionally dependent.

The presence of a fire ignition does not necessarily indicate a fire initiation. Consequently, the annual number of fire ignitions may or may not be the same as the annual number of fires. A fire initiation probability is therefore critical to determine the relationship between the numbers of fire ignitions and fire initiations (Li et al. 1997; Li 2000a).

Fire spread probability, however, may or may not be the same as the probability of fire initiation, because it is also a function of landscape topography and the operations of fire suppression. The methods of assigning values to the two fire probabilities in spatial fire regime models may also be different. Most spatial fire regime models assign a single value to the two fire probabilities for simplicity, and only a few models such as Ontario FIre REgime model (ON-FIRE) (Li et al. 1997) and SEM-LAND (Li 2000a) assign different values to the two fire probabilities. Mathematically, assigning a single value to the two fire probabilities is a special case of assigning different values to the two fire probabilities. Nevertheless, the two different fire probabilities would provide a greater flexibility for investigating the consequences of different scenarios, which is especially useful in simulating the influence of fire suppression on the dynamics of both fire regimes and forest ecosystems.

2.2.2. Fire Spread Simulation

A semi-stochastic fire spread process is a combination of stochastic influence and deterministic relationships that were summarized from fire behavior studies (e.g., in the Canadian FWI and FBP systems, see Van Wagner 1987 and Forestry Canada Fire Danger Group 1992). Two types of spatial models in simulating a fire spread process were reported in the literature: fire event simulators and fire regime simulators.

The fire event simulators are those mainly deterministic models aiming at reproducing a fire event in every detail by linking available dynamic weather variables to the formula of calculating fire perimeter locations continuously. The fire growth models are good examples of fire event simulators and have been paid a lot of attention historically due to their potential usefulness in guiding fire suppression operations.

The development of fire growth models in Canada such as Prometheus can be traced back to the model of Kourtz et al. (1977), which simulated a fire with a non-elliptical shape on a grid of cells (2 ha in size) representing a forest landscape. Fire spreads from one mid-cell point to an adjacent mid-cell point in a time interval determined by the fuel types of the two cells and their corresponding rates of spread. Since a fire can reach a cell's center from many different directions (i.e., not necessarily from the closest burning cell), the Dijkstra labelling algorithm was used to determine the shortest time-paths from the ignition cell. The model used a bookkeeping scheme to keep tracking the times required for fire to spread from cell to cell and thus the perimeter can be determined at any specified time.

The Huygens' principle, i.e., a wave perimeter can be propagated using any point on its edge as an independent source of a new "wavelet" (Anderson et al. 1982; Knight and Coleman 1993; Richards 1994; Richards and Bryce 1995), was also employed for simulating fire perimeter expansion. The vector-based FARSITE model (Finney 1994, 1998, 1999), for example, simulated the propagation of a fire perimeter as the expansion of a number of vertices. Elliptical fires described the new wavelets with a size determined by the fire spread rate determined by the local environmental conditions and a fixed time step. Local wind and slope would determine the orientation of the elliptical wavelets, and the length-to-breadth ratio determines the shape of each wavelet.

Many improvements in fire event simulators have been made such as the incorporation of the probability of encountering a rain that could be enough to stop a fire (Latham and Rothermel 1993; Anderson et al. 1998), improvement of the weather data input accuracy (Beck and Trevitt 1989), development of several length-to-breadth ratio (LB) equations to improve the fire growth algorithm (Alexander 1985; Rothermel 1991; and Richards 1994), long-range fire movement (Wiitala and Carlton 1994), and the incorporation of spot fires (Albini 1979, 1983), etc.

The major difficulty of using fire event simulators to investigate the long-term effect of fire suppression on fire regimes and forest ecosystems is the variability of reasons associated with why and how fires stop. For example, given enough time, a simulated fire would burn a forest landscape completely. Consequently, resulting fire size distributions could hardly be comparable to field observations. Nevertheless, two potential solutions are available: develop predictability for fire duration in probabilistic expressions, or incorporate stochastic effect on fire spread process as pointed out by Kourtz et al. (1977).

In contrast to the fire event simulators, the fire regime simulators are those aimed at estimating long-term dynamics of fire disturbances, and the temporal resolutions were usually coarser than those in the fire event simulators. The differences in temporal resolution also reflect that the simulation focus has been shifted from the perimeter locations of a fire at different time steps, to the occurrence and size-related descriptions such as annual fire number, area burned, fire cycle, fire size distribution, and spatial locations of individual fire events.

The algorithms of simulating a fire spread process in fire regime simulators include cellular automata (Wolfram 1985; Phipps 1989, 1992), percolation (MacKay and Jan 1984; Albinet et al. 1986; Ohtsuki and Keyes 1986; Von Niessen and Blumen 1986, 1988; Hargrove et al. 2000), and a special case of general epidemic processes (Grassberger 1983; Cardy and Grassberger 1985; Von Niessen and Blumen 1988). In cellular automata, systems of individual cells interact based on the "neighborhood coherence" principle. Any given cell tends to impose themselves on neighboring cells, and results in a tendency of local coherence and can display complex overall behavior (Wolfram 1985; Phipps 1989).

The percolation method focused on how random features at a small scale (e.g., a burning cell) would determine the overall behavior of a system based on the critical probabilities for a fire to become self-sustaining. In early applications of the percolation method, the research focus was to determine a critical probability under which a disturbance could spread from one edge of a landscape to the other edge (O'Neill et al. 1992). In the EMBYR model that simulated the effect of the spatial fuel class configuration on the spread of fire in the Yellowstone National Park, the number of critical probabilities was increased (Gardner et al. 1996; Hargrove et al. 2000). The fire spread has also been simulated as an application of general epidemic process that was closely related to the percolation method (Grassberger 1983; Cardy and Grassberger 1985; Von Niessen and Blumen 1988).

The influence of fuel quantity on the dynamics of fire regimes has been a topic of extensive discussion. Heinselman's (1973) assumption that increased fuel loads lead to increased fire behavior potential was supported by the observations reported by Minnich (1983), Riggan et al. (1988), and Turner et al. (1994). Van Wagner (1983) suggested that fire behavior in the boreal forest is a complex function of forest age, and that the potential fire intensity may not be proportional to fuel weight per unit area. For the Canadian boreal forests, it has been widely recognized that fire behavior is significantly influenced by weather conditions (e.g., Bessie and Johnson 1995); however, no fire regime descriptors were satisfactorily predicted directly from weather variables.

A systematic investigation of different fuel accumulation patterns on fire regimes has been reported in Li et al. (1997), where four fire probability functions including forest age-independent, hyperbolic, sigmoid, and linear increase with forest age were examined. An extensive model behavior study suggested that fire regimes similar to the observed temporal fire disturbance patterns in Ontario could result from any of the four fire probability functions, but with different sets of parameter values. Although the understanding of how the changing fuel quantity would influence the dynamics of a fire regime is still under development, it is obvious that the fire regime simulators can serve as appropriate tools to address the issue of long-term impact of fire suppression on forest ecosystems.

Fire suppression is one of the major human activities in fire management. The effect of fire suppression was also simulated differently in the two types of fire models. In the fire event simulators, detailed ground fire suppression activities were considered. In the AIRPRO model for air tanker operations (Simard and Young 1978), for example, free-burning fire growth until the start of fire suppression, perimeter growth of a fire during suppression, air tanker utilization, the final fire size when the fire is successfully suppressed, and the mop-up and patrol after a fire is suppressed were simulated. In order to simulate the perimeter growth of a fire during suppression, different equations were used to calculate fire growth in different directions, rate of fire control line construction was estimated based on observations, and thus total perimeter controlled by ground suppression could be calculated. However, the model was not intended to simulate fire growth if a fire escaped from fire suppression operations.

In the fire regime simulators, Baker (1995) estimated the two fire size probability distributions in the pre-settlement and fire suppression eras. The two distributions were then used as input of the DISPATCH model, to investigate the long-term consequences of these two fire regimes on forest landscapes. This modeling approach entirely relies on a sound knowledge of fire size distribution under fire suppression, without simulations of detailed fire suppression operations. The next section describes the method used in current study of C dynamics in boreal forests.

2.3. Simulation of C dynamics in boreal forests of western Canada

The boreal forests of western Canada have been historically shaped by fire regimes, so the C dynamics are also deeply influenced by fire disturbances. Therefore, simulations of fire regime and associated forest dynamics should not be separated. We employed the scenario-based Woodstock model (Walters and Cogswell 2002) and the process-based SEM-LAND model (Li 2000a) to simulate landscape C dynamics under different fire regimes.

With the stand-replacement fire-effect assumption, two main simulation algorithms need to be implemented in the models of forest landscape dynamics: inventory projection without disturbance and fire disturbance regimes. For the forest inventory projection without disturbance effect, the best quantitative relationships available are the results from regional forest growth and yield research. The forest growth equations for different species associations in Saskatchewan were used in both vector-based Woodstock and raster-based SEM-LAND models. The original forest inventory data are in vector GIS (Geographic Information Systems) format, and they were converted into raster format at a 1-ha spatial resolution for being used in the SEM-LAND model. In spite of diverse fire regime models reported in the literature (Keane et al. 2004), we implemented two different algorithms of fire simulation in the Woodstock and SEM-LAND models. We shall describe the study area and the details of models and model experiments in Section 3.

3. Materials and Methods

3.1. Study area

The Fort A La Corne (FALC) study area, with a total size of 132,502 ha, is a forested landscape surrounded by agricultural lands located in central Saskatchewan (Figure 1) (Saskatchewan Environment 1999). This area is within the Boreal Transition Ecoregion that represents the gradation from the grasslands of the south to the boreal forest of the north. Fine sands dominate the soils of the FALC. The major tree species are jack pine, aspen, black spruce, and white spruce. This area is about 500 m above sea level with a generally flat terrain. The FALC has 79,493 ha (60%) of timber producing land, 33,861 ha (25.5%) of productive non-forest, 18,038 ha (13.5%) of non-timber producing land and 1,110 ha (1%) of water. Fire has been the key natural factor that controls forest species composition and age structure and produces vegetation patterns. The forests in FALC are particularly susceptible to fire because of light rainfall and lack of moisture retaining soils, exposure to adjacent farmland and burning permit areas, long-term exploitation of forest products, abundant dry fuels, and infestation of disease such as dwarf mistletoe. Since 1943, most fires in FALC have been relatively small except for three: the Steep Hill fire of 1967 burned 13,700 ha; the Henderson fire of 1989 burned 11,100 ha; and the English fire of 1995 burned 28,500 ha. These large fires have substantial impacts on forest harvest, forest renewal, and wildlife habitat. The AAC in the area is 57,055 m^3 of softwood and 37,628 m^3 of hardwood. The current harvest level is 28,855 m^3 of softwood and 8,500 m^3 of hardwood. The concept of emulating natural patterns of fire in harvesting has been considered in the research and education plan.

In the model, current forest conditions in the area were represented by GIS data layers such as the forest cover type, stand age, and tree density at 1-ha resolution (Fig. 2).

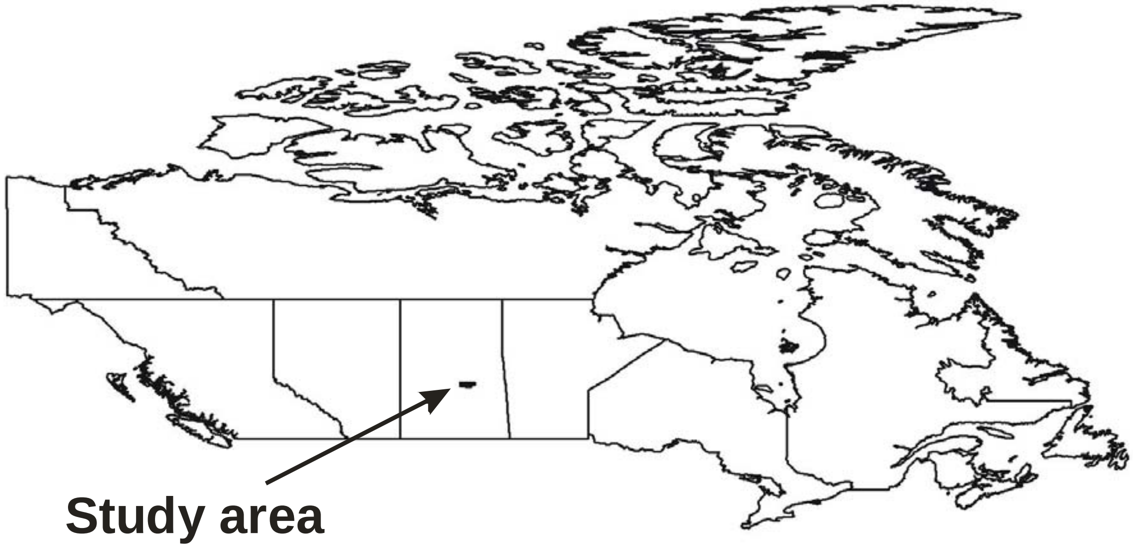

Fig. 1. Location of the study area in central Saskatchewan, Canada.

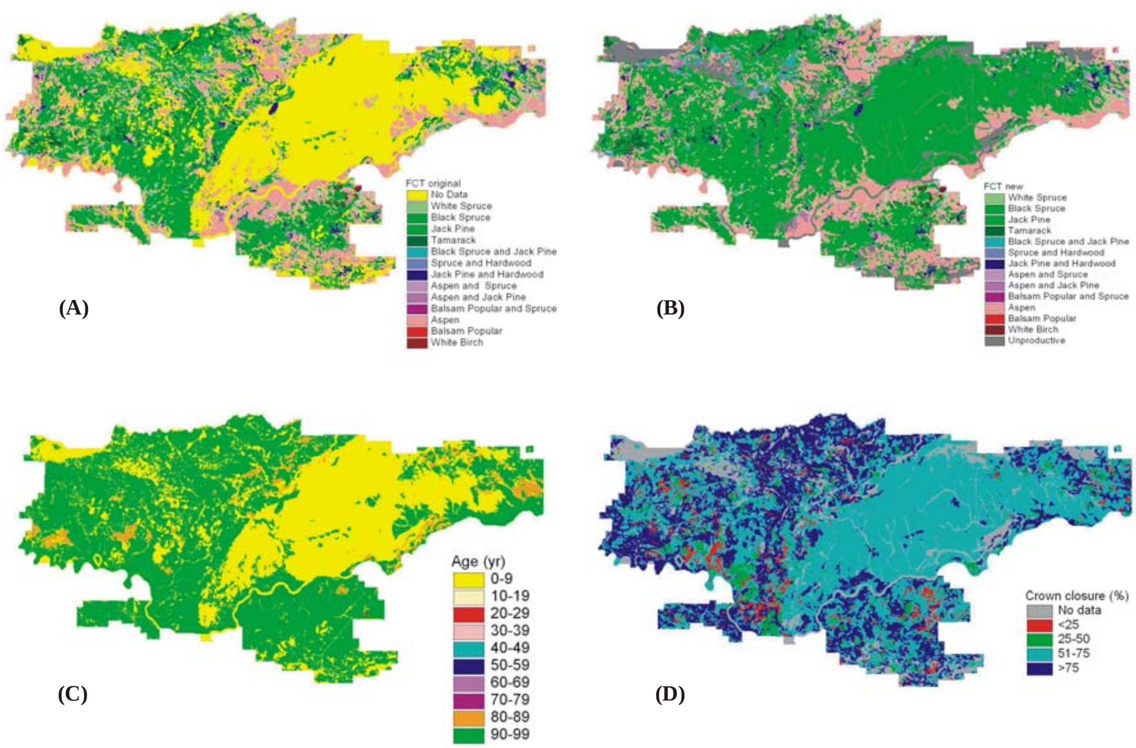

Fig. 2. Maps of the Fort A La Corne study area, Saskatchewan. (A) Original land cover type. (B) Reconstructed land cover type. (C) Forest age. (D) Stand density.

Forest inventory records were not available for about 40% of the total FALC area, because several large fires had occurred in 1995 including the English fire event (Fig. 2A). This created a difficulty with the fire simulations because the burned area almost separated the FALC into two parts, and thus a fire started in one part could hardly spread to the other. To provide this missing data, a map of provincial fuel types was placed over the forest cover type layer, and the corresponding forest cover types were converted for those pixels with missing data. This provided a reconstructed forest cover map for the requirement of the model simulations (Fig. 2B). For these pixels, we set the stand age for the burned area to zero (Fig. 2C), and tree densities to an intermediate level C that is 51-75% of crown closure (Fig. 2D). These reconstructed layers might contain bias in estimated forest productivity but the qualitative conclusions drawn from this simulation investigation should not be affected.

The forest age distribution was determined by using the FRAGSTATS software package (McGarigal and Marks 1995). A time-since-fire distribution was constructed from the forest age distribution according to Johnson and Gutsell (1994) following a methodology review (Li 2002). This data was used to determine whether the fire regime had changed in the past. Li (2002) showed that stand-origin map-based estimation of fire cycles could overestimate the length of a fire cycle. This could be particularly exaggerated in areas burned recently by large fires. Therefore, the stand-origin map of the study area portion that was not burned by the large fires in 1995 was used for the fire cycle estimate.

3.2. Model descriptions

Two models were used in our modeling investigation. One is the commercial software package, Woodstock, designed for operational timber supply analysis and harvest planning. The other is the process-based spatial fire regime model SEM-LAND.

3.2.1. Woodstock model

Commercial software packages for timber supply analysis are available and widely used in various resource management agencies across Canada for harvest planning. For example, Woodstock and Stanley of the Remsoft (Walters and Cogswell 2002) are used in Alberta, Saskatchewan, Manitoba, Northwest Territories, and Atlantic provinces. Non-commercial software packages developed by researchers in other provinces are also used for timber supply analysis. For example, Ontario uses the strategic forest management model SFMM and the sustainable management optimization model PatchWorks (Spatial Planning Systems 2004). In B.C., the forest service simulation model FSSIM is used for timber supply analysis (BC Ministry of Forests 1996). Due to the wide application in western Canada, Woodstock is appropriate for our current modeling investigation.

Regardless of the differences in the software packages, the methodologies of estimating timber supply are essentially very similar. For instance, the Woodstock inputs forest inventory data in GIS format to calculate timber supply over a multiple planning period based on user-defined yield equations for different tree species or species association. The calculation is then optimized through linear programming to allow relatively constant AAC determined for different periods of harvest planning. This relatively constant AAC is ideal for the stable timber production and processing industries.

A disadvantage of using the Woodstock model in the current study is that the model itself only considers how the timber resource could be utilized in an optimized manner based on normal forest growth. Other disturbances such as fire are not incorporated into the Woodstock model. In the modeling investigation of how C dynamics could be influenced by fire and harvest regimes, we assumed that the burned area in a given forest type is proportional to its total area within the landscape. Therefore, the estimated annual area burned from a given fire regime can be allocated to different types of forests. This assumption may or may not hold in different regions. Nevertheless, this algorithm provided an approximation of the fire effect on AAC determination.

The spatial allocation of the harvest blocks within a forest landscape under management that best suits operational and strategic goals and constraints can be achieved by using other software packages such as the Stanley and PatchWorks models. This allocation may not directly influence the determination of AAC, but the post-harvest landscape structure could have significant impacts on non-timber values and disturbance risks, thus on the sustainable regional development. Dynamic interactions among AAC determination and spatial allocation of harvest blocks over time are thus not realized in this modeling environment, but this implementation strategy is probably the best utilization of the commercial software package. To consider the effects of the interactions among all these factors on the same forest landscape, it would be ideal to have major disturbance regimes including fire, MPB, and harvest regimes simulated in a single model. However, such an integrated disturbance model is still under development and we shall discuss it further in the Section 6.

3.2.2. SEM-LAND model

Since a detailed SEM-LAND model description on the assumptions, structure, quantitative relationships, and its applications in addressing different issues has been documented in a series of publications (Li 2000a, 2002, 2004; Li and Apps 2004, Li et al. 2000, 2005a, 2005b), only a brief description will be presented here.

The SEM-LAND model simulates natural fire regimes based on current forest landscape conditions represented by four map layers of a study area including land cover, elevation, density, and forest age in central Saskatchewan. Fire disturbance simulation is one of the major components of the SEM-LAND model. Two baseline fire probability maps (for initiation and spread) were constructed from a current forest age mosaic pattern, and the maps were modified by a number of scale factors as follow:

$$P_{Initiation} = P_{Baseinitiate} \times F_{WeatherFuel} \qquad (1)$$

$$P_{Spread} = \begin{cases} 0 & (R \geq R_{Crit}) \\ P_{Basespread} \times F_{WeatherFuel} \times F_{Slope} \times (1 - \mathrm{FSE}) & (R \,\pi\, R_{Crit},\ S \,\pi\, S_{Crit}) \\ P_{Basespread} \times F_{WeatherFuel} \times F_{Slope} & (R \,\pi\, R_{Crit},\ S \geq S_{Crit}) \end{cases} \qquad (2)$$

where $F_{WeatherFuel}$ is the scale factor calculated according to the Canadian Forest Fire Behavioral Prediction system (FBP) (Forestry Canada Fire Danger Group 1992) representing the influence of fuel type and weather conditions, F_{Slope} is the scale factor due to slope, R is the daily precipitation, R_{Crit} is the critical value of daily precipitation and any precipitation that reaches or exceeds the value can stop a fire, *FSE* is the fire suppression efficiency, S_{Crit} is the critical value of fire size and any fire that reaches or exceeds the value can escape from fire suppression, $P_{Baseinitiate}$ and $P_{Basespread}$ are the baseline fire probabilities for initiation and spread stages, and they are characterized by a logistic equation:

$$P_{Base} = k / (1 + \exp(a - b \times Age)) \qquad (3)$$

where *a*, *b*, and *k* are parameters, and *Age* is the forest stand age. This logistic equation describes the possible patterns of fuel accumulation since last burn, assuming that the fuel quantity is a function of forest age.

A fire event is simulated in two stages: fire initiation and spread. In the fire initiation stage, fire ignition locations are randomly determined. Once a fire ignition location is determined, the program will calculate the baseline fire initiation probability and scale factors using the land cover, hence the fuel type, the Fine Fuel Moisture Code (FFMC), and the time since last burn. The FFMC was determined through a random sample from the daily FFMC frequency distribution based on historical weather data. A uniformly distributed random number will be generated and compared with the resulting fire initiation probability to determine whether a fire initiation would result.

The fire spread stage starts when a fire is initiated at its ignition location, the program will then check its potential to spread into its eight adjacent cells based on the fire spread probabilities and scale factors calculation for each of adjacent cells. Again, a uniformly distributed random number will be used to compare the resulting fire spread probability to determine whether the fire would spread to the given adjacent cell. If the cell were burned due to its adjacency to a burning cell, then another round check of its eight adjacent cells would start. Under the fire suppression situation, the equation selection for fire spread probability calculation was dependent on the fire size cumulated and the daily precipitation. The fire would stop when the daily precipitation exceeds the R_{Crit}, or stop in the direction where encountering a non-flammable or a low flammable cell or when reaching the boundary of the study area. A cell could be burned from eight directions, and consequently, the final fire shape might not be regular, and unburned remnants could appear.

A self-replacement of species composition was assumed and the volume of burned cells would change accordingly. This assumption was to reflect the observations that after about 30 years or so, the species compositions in Canadian boreal forests would come back to the one before the stand-replacement fires. However, diverse species compositions could appear for about the first 30 years. The volume growth equations of major forest cover types from Saskatchewan Environment were used for our current study (Li et al. 2005). The SEM-LAND model behavior was improved through the replacement of a random wind direction with a frequency distribution of wind direction from historical data statistics summarized in Wolfe and Ponomarenko (2001), the inclusion of the effect of weather conditions on growth rates of major tree species from Sauchyn and Beaudoin (1998), and the incorporation of fire suppression simulation that was abstracted from the conceptual understanding of fire suppression processes.

A lightning fire process under fire suppression is simulated in three stages. Stage one starts when a fire ignition occurs from a lightning strike at a site on forested land, until it has been detected, reported, and fire crews arrived on the scene. During this stage, the fire initiates and spreads freely without human intervention, i.e., under natural conditions. With the increasing capability of wildfire detection and the advancement of transportation equipment such as helicopters, the duration of this stage became shorter. The second stage is from the beginning of the initial attack, until the fire has been stopped by management operations or the fire has escaped from the fire suppression effort. The fire spread probability would be reduced by fire suppression efforts during this stage. The third stage is from the beginning of a fire escaping from a fire suppression effort, until it is finally stopped. In this stage, the fire spread process would be unconstrained again, i.e., the fire spread probability would revert to the one representing natural conditions. It should be noted that this three-stage fire suppression description was not intended to capture every detail of ground fire suppression operations, but an abstract that could be implemented in the fire regime simulators for the investigation of possible long-term consequences of fire suppression on the dynamics of fire regimes and forest ecosystems.

The three-stage fire suppression was simulated in the SEM-LAND model through the effect of combined *FSE* and S_{Crit}. The *FSE* was to indicate the level of protection executed by the fire suppression agencies, which was determined by the resource investment to the operation and the equipment and facilities used in the operations. The *FSE* was indicated by a reduced percentage of fire spread probability due to fire suppression. The S_{Crit} indicated a starting point when the fire spread probability would return to the level representing natural conditions.

3.3. Model experiments

The purpose of this model investigation is to explore the forest C dynamics under all possible fire and harvest regimes from a long-term perspective. This has resulted in the model experimental design in which we included complete ranges of possible AAC levels and fire cycles.

The Woodstock model was run for investigating forest C stock dynamics under different scenarios: (1) without the effect of fire and harvest disturbance regimes; (2) various AAC levels (50000, 75000, 100000, 125000, 150000, 175000, and 190000 m^3) without fire disturbances; (3) various fire regimes (fire cycles of 50, 75, 100, 125, 150, 200, 300, and 400 years) without harvest regimes; and (4) different combinations of fire (fire cycles of 50, 75, 100, 125, 150, 200, 300, and 400 years) and harvest (AAC of 50000, 75000, 100000, 125000, 150000, 175000, and 190000 m^3) regimes.

Each simulation lasted 200 years with every 10 years as a planning period. Under each scenario, forest age and volume in each forest type at different planning periods were recorded. Ten replications were performed for each of the model simulation scenarios.

The SEM-LAND model was run for estimating forest C stock dynamics under natural fire regimes in the FALC.

3.4. Data analysis

The simulated forest volume dynamics are converted to C contents using the method summarized in von Mirbach (2000) that was prepared for Canada's Model Forest Program on C accounting at the forest management unit level. The conversion factors are based on the IPCC's Greenhouse Gas Inventory Guidelines Reference Manual and modified by Environment Canada using Canadian data. They are considered acceptable for national and international reporting requirements. The aboveground forest volume (m^3) is estimated by multiplying the merchatable volume by a factor of 1.454. The below-ground volume is estimated as 0.396 of the aboveground volume. The total wood volume is then converted to dry matter biomass (tonnes) by a factor of 0.43. The C stock is considered as one half of the dry matter biomass.

For comparitive purpose, C stock is also estimated by using the IPCC default method (Penman et al. 2003) and CanFI data. In the IPCC default method, forest growth is expressed as MAI for a given forest type in given region. For Canadian boreal forests, the average annual increment in mixed broadleaf-coniferous aboveground biomass is estimated as 1.1 tones dry matter/ha/year. For coniferous, the estimate is 0.8 for forests younger than 20 years, and 1.5 for forests older than 20 years. The biomass expansion factors to include below-ground biomass are 1.35 for boreal conifers and 1.3 for boreal broadleaf. Biomass consumption for boreal forests is 52.8 tonnes/ha in general. In more detailed consideration, 25.1 t/ha for crown fire, 21.6 t/ha for surface fire. The combustion factor values (proportion of prefire biomass consumed) are 0.40 in general, 0.43 for crown fire, and 0.15 for surface fire.

The boundary of the FALC area is overlaid with CanFI cells. The CanFI data have a spatial resolution of 100 km^2 per cell. Twenty-six CanFI cells are identified that are related to the FALC. These cells can be fully within the FALC or as little as 0.12% within the area. All the 971 records for individual stands were collected for analysis. We then added all the forest volumes together for estimate C stock within the FALC.

Results show that total volume is 7,154,040.239 (m^3) and 3,076,237.303 tons of dry biomass. This equivalent to 1,538,118.651 tons of C at the time of data collected.

4. Results

We performed a model investigation using the Woodstock model to explore the C stock dynamics under different combinations of the fire and harvest regimes. We converted the forest inventory data into raster format at a one-ha resolution. Simulations were run for 200 years with a planning period of 10 years. Ten replications were performed for all the combinations of fire and harvest regimes except the scenario of no disturbance, where only one simulation was performed.

4.1. C stock dynamics without disturbance

The first scenario we examined was of no disturbance over time in which forests could grow normally and regenerate when reaching their longevity (200 years). This provided an estimate on whether C stock could continue to increase when the complete forest protection is achieved. Figure 3 summarizes the simulation results. The Figure 3A shows that the mean forest age can be increased gradually until 180 years old at the simulated 120 years and then decline. Figure 3B indicates that the mean forest C stock per hectare can be increased in the first 50 years and then gradually decrease for the following

100 years. After that, the C stock will increase again to recover the initial C stock level at the last planning period. The different dynamic patterns in mean forest age and mean C stock are essentially because the mean annual increment (MAI) of volume is not constant over time as indicated in Figure 3C. We also examined the dynamics of areas older than 120 years (Figure 3D) and found a similar pattern with mean forest age (i.e., the old forests can increase until 100 - 110 years and then decrease).

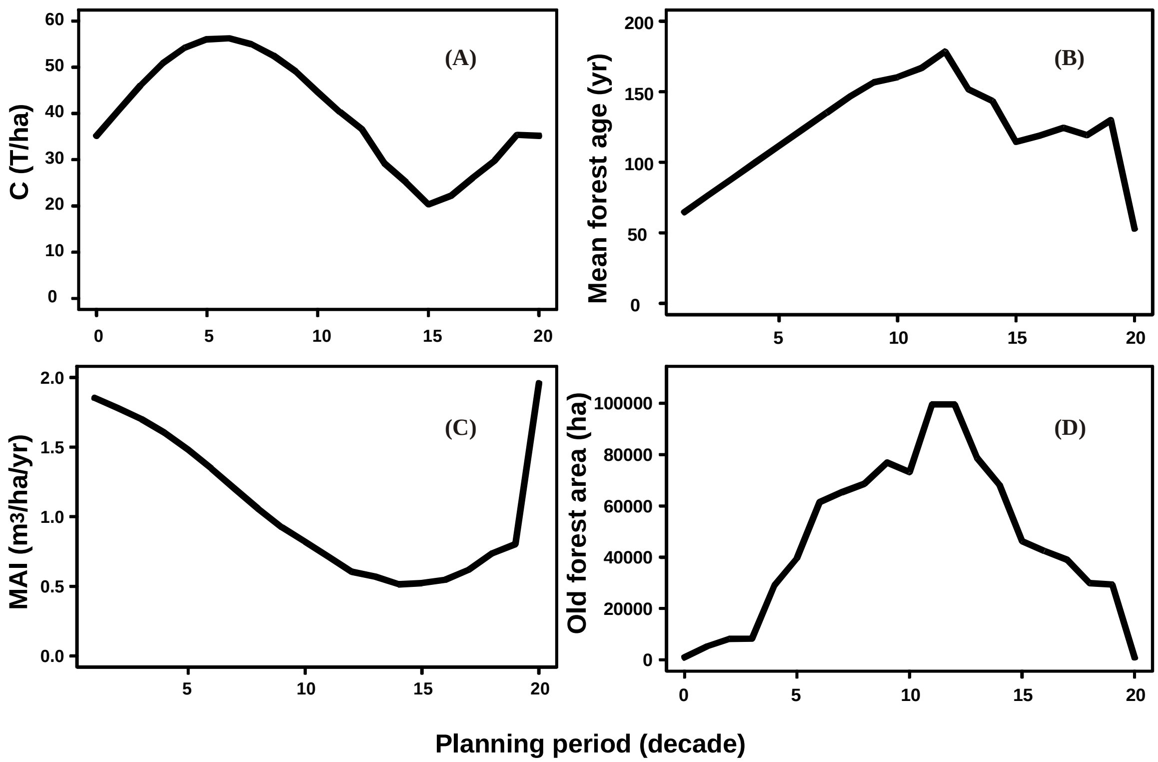

Figure 3. Forest and C stock dynamics without disturbances. (A) Mean forest age. (B) C stock per hectare. (C) Mean annual increment (MAI). (D) Old forest area.

Our simulation results indicated that when forests have a complete protection, forest C stock cannot increase without a limit because forests can only grow until their longevity is reached. From a long-term perspective, therefore, complete protection may not be the best strategy to increase C stock, as well as the old forest area.

4.2. C stock dynamics under different AACs

We then investigated how the forest C stock could change under different harvest rates expressed as AAC. This is similar to standard timber supply analysis and does not take into account of fire disturbance effects. We simulated forest and C dynamics using the Woodstock model with the AACs of 50000, 75000, 100000, 125000, 150000, 175000, and 190000 cubic meters (m^3) to explore the range of AAC that a sustained timber supply could achieve in this study area. An exploration of the maximum AAC was also included. Figure 4 summarized the simulation results. From Figure 4A (only the AACs of 50000, 100000, 150000, and 190000 m^3 are presented), we can see that the patterns of mean forest age change over time are similar, but the highest mean forest age will decrease with increasing AAC. The fluctuation pattern of total volume (Figure 4B) is similar to the mean C stock in Figure 3B, but the amplitude of fluctuation will decrease with increasing AAC. When examining the volume change, a decreased value can be expected when the AAC becomes larger (Figure 4C). The old-forest area dynamics (Figure 4D) also show a similar pattern in comparison with the no-disturbance scenario, but the highest value for the old forest area will also decrease with an increasing AAC.

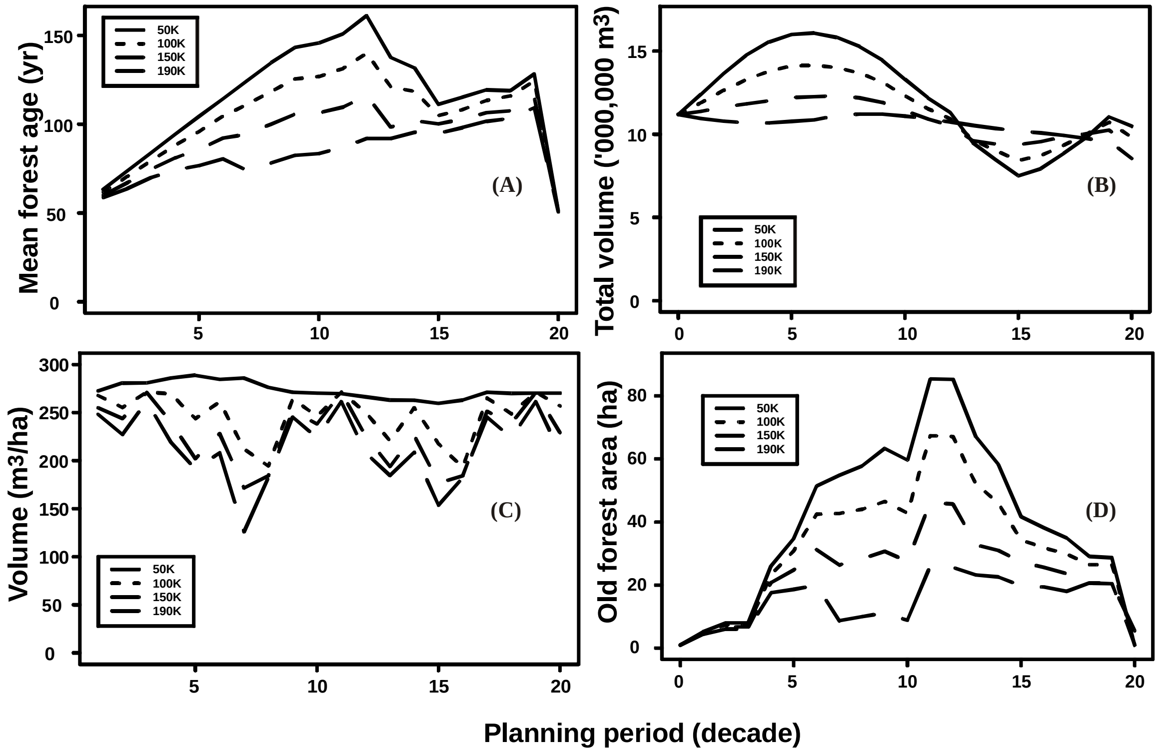

Figure 4. Forest and C stock dynamics under different annual allowable cuts. (A) Mean forest age; (B) C stock per hectare. (C) Mean annual increment. (D) Old forest area.

Any simulations with an AAC larger than 190,820 m^3 cannot reach 100% success (i.e., to generate a solution of a sustained timber supply). This means that the 190,820 m^3 is very close to the maximum harvest level, or annually allowed stand-replacement disturbances, that the forests in the study can sustain.

We also summarized the mean values over 20 planning periods and 10 replications to show the trends of forest and C stock dynamics under various AACs in Figure 5. The trends can be summarized as follows: mean forest age (Figure 5A), total forest inventory (Figure 5B), volume per hectare (Figure 5C), and old forest area (Figure 5D) will all decrease with increasing AAC.

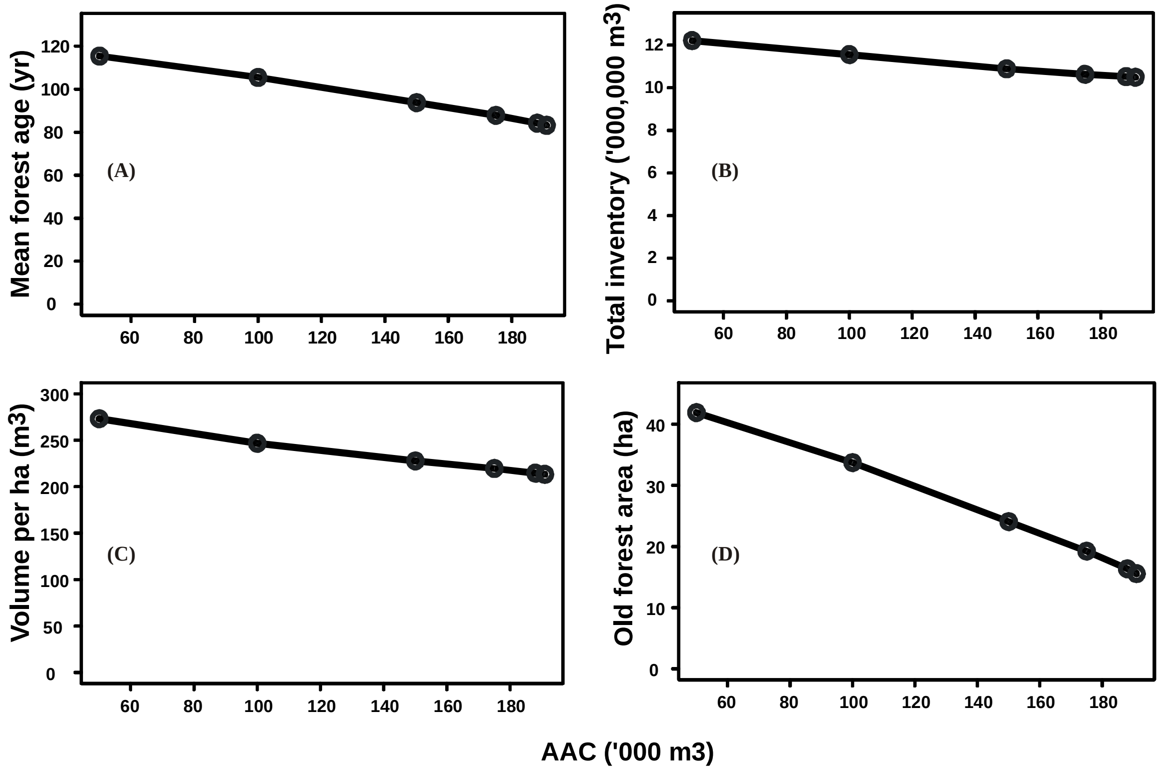

Figure 5. Forest and C stock dynamics under different annual allowable cuts. (A) Mean forest age. (B) Total volume. (C) Mean volume per hectare. (C) Old forest area

4.3. C stock dynamics under different fire regimes

The Woodstock model was also used to simulate forest dynamics under different fire regimes, expressed as the fire cycle. Harvest effect was not considered in this model experiment by setting the AAC to zero. A given fire cycle means a certain percentage of forest land will be burned annually, and the annual burned area was allocated among available areas of different forest types. We ran the model 10 times under a fixed 100-year fire cycle first, and the simulation results are summarized in Figure 6. Despite small variations among the 10 simulation replications, the mean value can capture the general trends in total inventory (Figure 6A), mean forest age (Figure 6B), and old forest area (Figure 6C). Figure 6A showed that the total inventory slightly decreased over the 200 years and this is consistent with Figure 8G where mean size of C sink decreases with short fire cycles. Figure 6D shows how the MAI of forest volume is related to mean forest age. It illustrates that the highest MAI is at a little younger than 60 years, and the MAI will be decreased with the increasing mean forest age. Consequently, maintaining mean forest age in a relatively younger stage could obtain higher MAI. However, mean forest age that is too young might not reach a higher MAI. Since Figure 6D decomposed Figure 6A in terms of MAI at different forest ages, it illustrated that the total inventory decrease over planning period was because the decreased MAI due to increased mean forest age (Figure 6B).

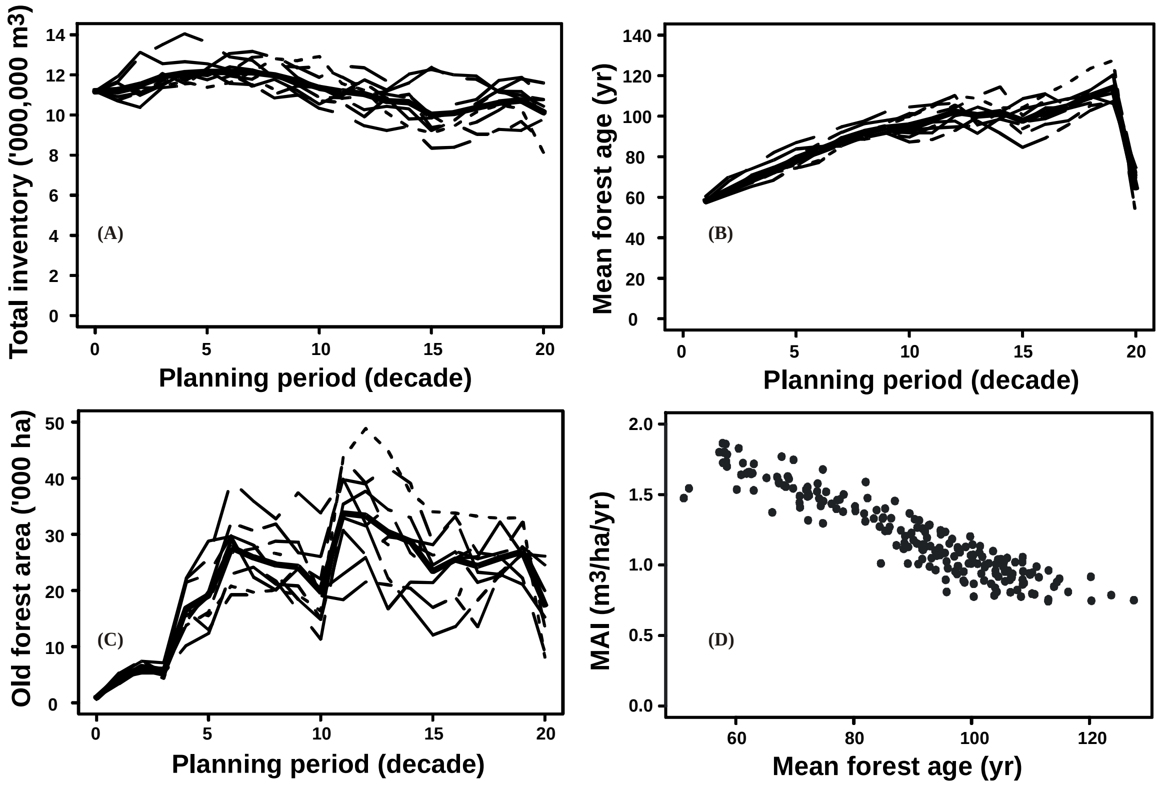

Figure 6. Forest and C stock dynamics under a 100-year fire cycle. (A) Total forest inventory. (b) Mean forest age. (c) Old forest area. (d) Mean annual increment (MAI) at different mean forest age.

We then performed a model experiment by running the Woodstock model under the fire cycles of 50, 75, 100, 125, 150, 200, 300, and 400 years, using the same method of allocating burned areas among available forest types. Figure 6 shows the results from this model experiment.

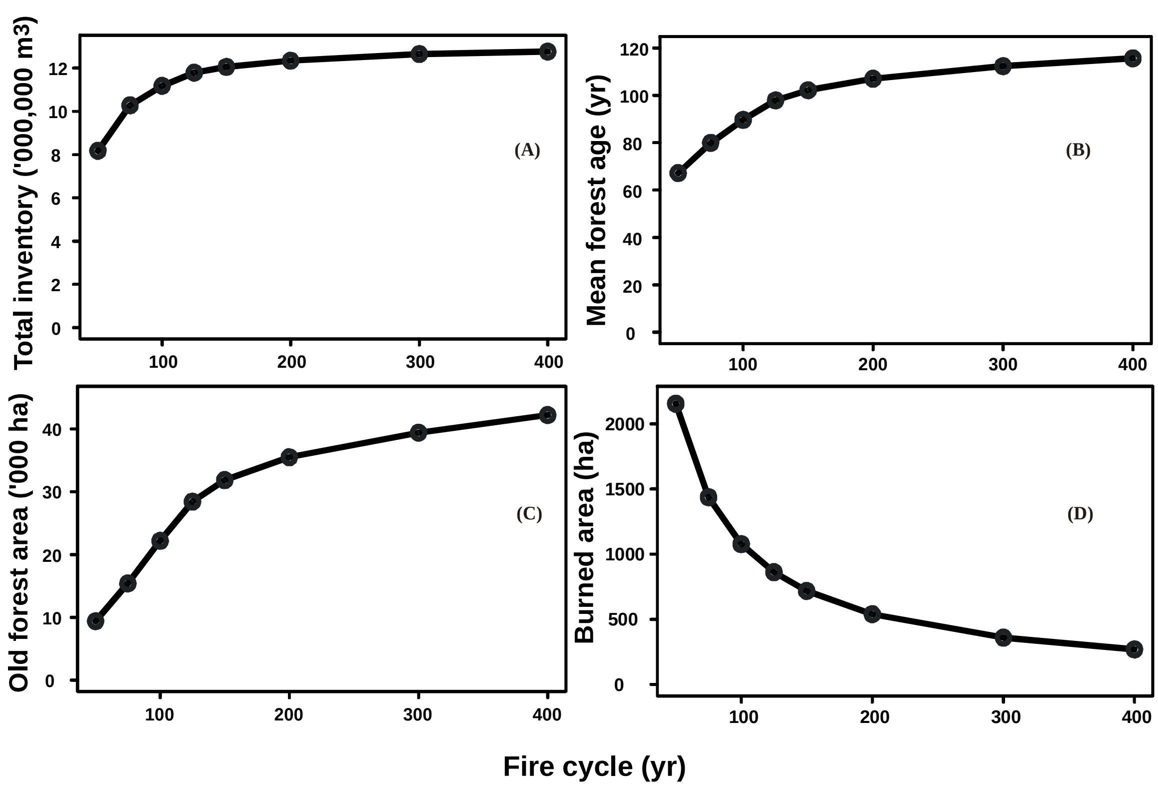

Figure 7. Forest and C stock dynamics under different fire regimes. (A) Total mean forest inventory. (B) Mean forest age. (C) Old forest area. (D) Mean area burned.

Our results indicate that the total mean inventory, mean forest age, and old forest area (across 20 planning periods and 10 replications) can rise with increased length of fire cycle (Figure 7A, 7B, and 7C), because the mean burned area (Figure 7D) decreased with increasing fire cycle length. These results have trends similar to those from other spatial fire regime models such as those reported using SEM-LAND simulations (Li 2004; Li et al. 2005).

4.4. C stock dynamics under different combinations of fire and harvest regimes

From an understanding of the trends generated by the simulated data under the three scenarios presented above, a systematic model experiment was performed to simulate forest dynamics under different combinations of fire and harvest regimes.

Not all of the combinations generated meaningful results because some simulations were infeasible before completing the 20 planning periods. Figure 8A shows the success rate of the simulations. High AAC and short fire cycles tended to contribute to the unsuccessful simulations. Therefore, all the simulation results from this experiment will contain missing values in the unsuccessful simulations of the combinations of fire and harvest regimes.

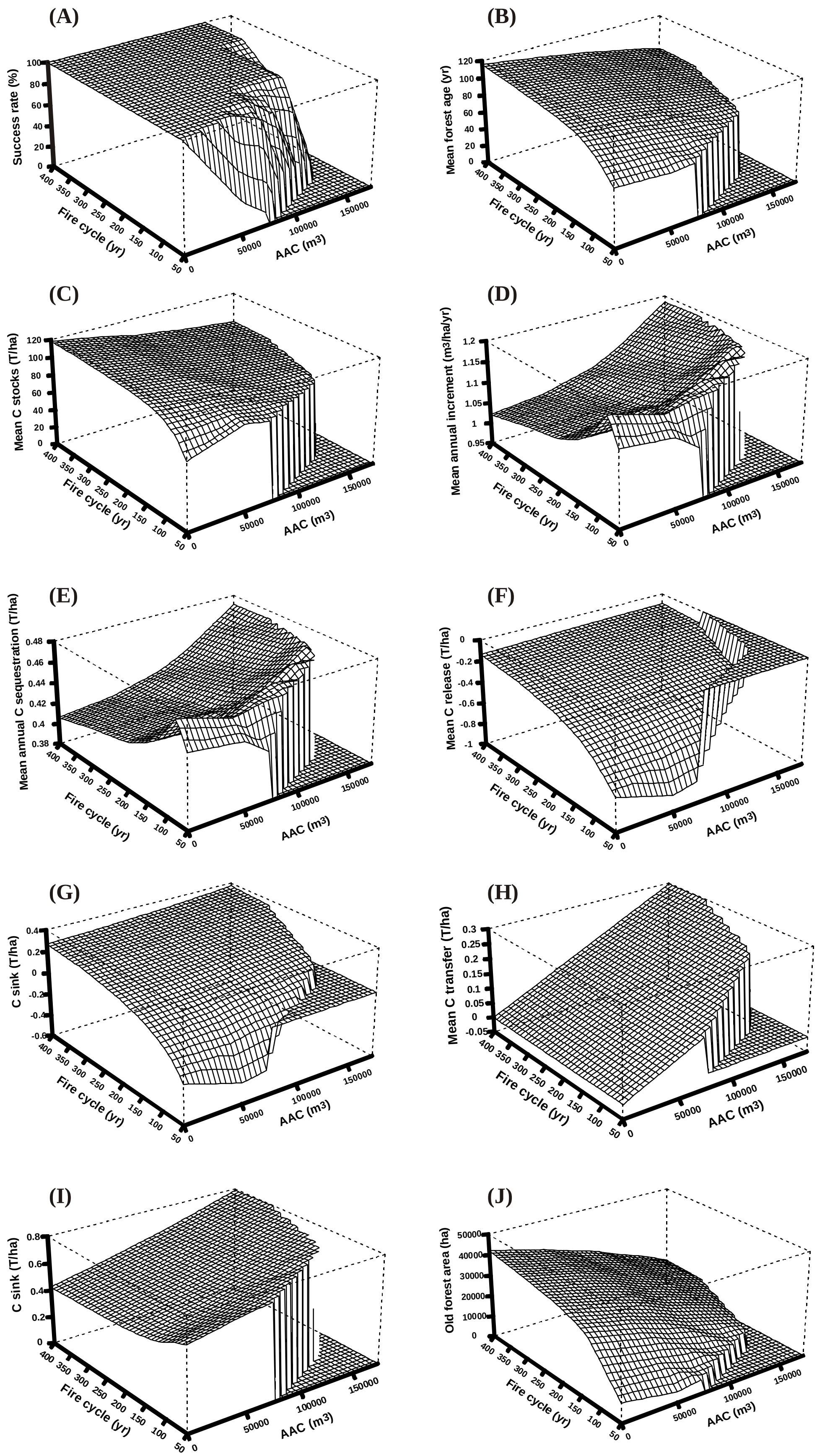

Figure 8. Simulated forest and C dynamics under different combinations of fire cycles and annual allowable cuts. (A) Success rate of simulations. (B) Mean forest age. (C) Mean C stock. (D) Mean annual increment (MAI). (E) Mean annual C sequestration. (F) Mean C release. (G) Mean C sink size. (H) Mean C transferred to product pool through harvest. (I) C sink size including those transferred to product pool. (J) Old forest area.

Figure 8B showed that mean forest age will be the highest with no harvest and the longest fire cycle. It will drop with decreasing fire cycle and increasing harvest rates. The mean C stock per hectare (Figure 8C) will have a similar surface with mean forest age, suggesting a positive correlation between the two variables.

MAI increased with disturbance rate, and this was true for both harvest and fire regimes (Figure 8D). This is because any stand-replacement disturbance can reduce the mean forest age thus pushing the MAI into a higher value (see Figure 8D). The MAI surface can be converted into a mean annual C sequestration surface (Figure 8E).

Figure 8F shows the surface graph of mean C release from the fire disturbances. This surface is generated based on the CBM-CFS2 method (von Mibach 2000). For the IPCC approved default values, the surface suggests a lower mean C release, essentially because of the different rates assumed. In the IPCC approved method, crown fires in boreal forests will have a 0.43 combustion factor defined as the proportion of pre-fire biomass consumed, and a 0.15 factor for surface fires. Thus, the average value of 0.29 can be assumed in the calculation. In CBM-CFS2, the factor 0.309 was used (Kurz et al. 1992). Based on 12345 historical fire records in Alberta during a period from 1961 to 1995, the crown fire burned 56.3% of total burned area on 38.3 million ha of Alberta's forest land, while surface fire burned 43.4% (Li 2004). In this case, using a combustion factor of 0.307 might be more appropriate, and we could expect the mean C release is somewhere between the results using IPCC approved method and CBM-CFS2 method for our study area.

Figure 8G shows the mean C sink size under various combinations of fire cycles and AACs. This surface is generated by subtracting C release (Figure 8F) from sequestration (Figure 8E). This surface represents a mix of positive and negative C sink sizes. Short fire cycles tend to make the C sink size negative thus representing a C source.

Harvest also transfers C from biomass to forest products and Figure 8H shows the mean transferred C under different AACs. If this transferred C is considered in the C sink size, the mean C sink size surface will look like Figure 8I, i.e., all positive values will appear in various combinations of fire cycles and AACs. Figure 8J is the old forest area size.

All the fire-released C estimates presented in Section 5 are the means across all the planning periods and replications. This treatment might not be able to capture some of the exceptional conditions such as the period with the highest forest growth rate, during which C sequestration is high, and the extreme fire disturbance conditions in which the amount of C released is very high. The understanding of C dynamics under these exceptional conditions, nevertheless, will provide opportunities for forest managers to design the best strategy for taking advantage of the highest forest growth-rate periods and avoid extreme fire disturbance conditions. This is particularly meaningful if this is linked to current and future forest management strategies. In the next section, we shall describe one of the modern forest management paradigms – emulation of natural disturbance (i.e., fire) patterns in harvest planning and how to reconstruct the natural fire patterns based on current forest conditions, as well as to estimate (1) the period in which forest growth rate could reach a level that is higher than the IPCC default estimate; and (2) the frequency of the extreme fire disturbance conditions if the emulation of natural fire patterns in harvest planning is implemented and the consequence in the C dynamics.

4.5. Emulation of natural fire patterns

Utilization of existing forest resources aims to satisfy human needs and economic activities. However, it is a concern that historical timber harvest has changed forest conditions so that future resource development might be significantly influenced for future generations. Based on the understanding that forest fire has historically shaped the dynamics of boreal forest landscapes, forest managers and researchers believed that emulating natural fire patterns in harvest planning would help to achieve the goal of sustainable forest resources and biodiversity maintenance. In Ontario, the Crown Forestry

Sustainability Act requires timber harvest to emulate natural disturbance patterns (Li 2000a). This forest policy has also attracted great interest in other provinces of Canada as well as some of the US States and has become one of the major forest management paradigms (Kimmins 2004). It is expected that this forest policy will become more popular in modern forestry practices.

Understanding natural fire regimes is important in harvest planning. Most fire regime observations have been made on managed forests; however, observed fire patterns might or might not be the same as natural fire patterns because of increased human influences such as decades of fire suppression operations. Thus, data on natural fire regimes need to be found or reconstructed through appropriate methods. The ecological modeling approach can provide alternatives to such data, if necessary natural processes are represented in the models reasonably well. However, the inclusion of known processes into a simulation model often results in a very complicated model design and hence demands considerable resources in estimating the large number of parameters and variables, and powerful computing facilities to complete the simulations in a reasonable time. FIRE-BGC (Keane et al. 1996) is one such model that has incorporated most known processes with more than 1000 parameters to estimate model input. Another challenge for such complex models is the assessment of the interactions among different processes. Positive and negative influences might generate results that are difficult to verify with observations. Consequently, models with a mid-level of complexity and containing only key processes will probably be the most suitable. The SEM-LAND model was originally designed for this purpose and was capable of simulating natural fire regimes based on current forest conditions as described in Section 3.2.2. The SEM-LAND model was capable of simulating historical and current fire regimes in the FALC (Li et al. 2005). In the current study, we use this model further to simulate forest C dynamics that are associated with natural fire regimes.

Figure 9A shows the simulated annual C sequestration, and Figure 9B denotes the simulated annual C released by fires. Figure 9C is the C sink size resulting from the sum of the C sequestration and release. Figure 9D shows the positive value portion of the Figure 9C. As Bhatti et al. (2004) pointed out the reason that boreal forests tend to be C sources is because of lower growth rates. Therefore, any forest management options that can enhance forest productivity will enhance C sink size. This can be identified by a comparison between the forest C sequestration under different management options and the C sequestration calculated by the KP default estimate. This can be seen clearly in Figure 9D: since the KP default estimate for boreal forests is about 0.46 t/ha, a forest age younger than 80 years old would probably result in a C sink size larger than 0.46 t/ha. Therefore, any forest management operation that can result in a mean forest age of younger than 80 years could be potentially good for enhancing C stock, and the C sequestration could reach the highest peak when the mean forest age is about 40 years. Considering the best utilization of wood materials, current harvest activities mostly focus on forests of 60-70 years. This study suggests that the forest rotation period could be shortened to obtain the highest C stock.

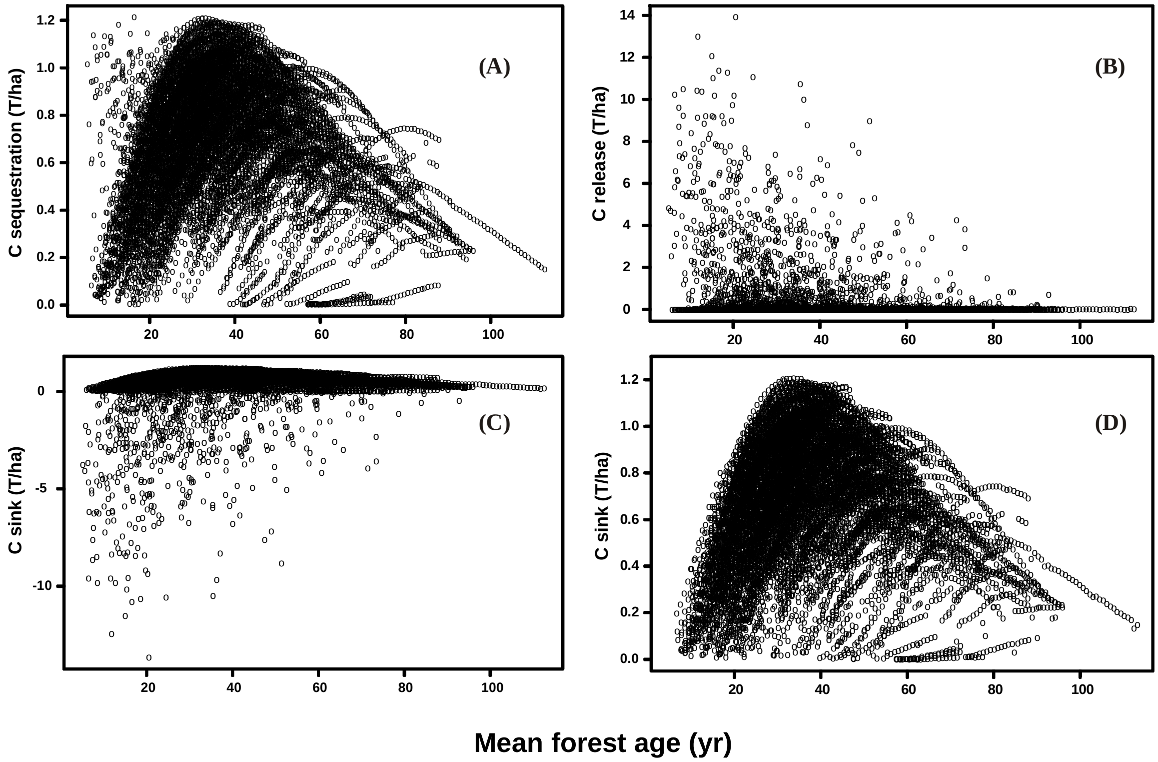

Figure 9. Forest C dynamics simulated by SEM-LAND model. (A) C sequestration by growth. (B) C release by fire. (C) C sink size. (D) The positive portion of C sink size.

5. Discussion

5.1. Role of forests in global C budget:

The role of terrestrial ecosystems in the global C budget has been extensively discussed in the literature (e.g., Watson et al. 2000; Ciais et al. 1995; Houghton 1996). A number of estimates at national or regional scales have concluded that temperate and boreal forests are C sinks (e.g., Kauppi et al. 1992; Sedjo 1992; Dixon et al. 1994) while other studies have reported that the forests function as a C source (e.g., Harmon et al. 1990). Kurz and Apps (1999) suggested that the role of forests in the C budget could change over time depending upon changes in the prevailing disturbance regimes. For instance, the forest ecosystems in Canada have been a sink of atmospheric C for the period 1920-1980, and a source for the period of the 1980s because of a sharp increase in forest fire and insect disturbances starting about 1970. The disparate results can be attributed to different geographical regions, climate and weather patterns, vegetation and soil conditions, disturbance regimes, and different methods of estimating C stock.

Brown et al. (1996) estimated that in high-latitude forests, about 2.4 Pg (1 Pg = 10^{15} g or billion metric tons) of C could be potentially sequestered and conserved by forest management practices from 1995 to 2050, and this is equivalent to about 0.46 tones of C sequestered and conserved per hectare per year. For Canada, 15 Pg of C was in vegetation and 76 Pg of C in soils, which resulted in a 0.08 Pg yr^{-1} of C sink.

Brown (1997) reported that in high-latitude forests, the total C pool is about 278 Pg and 71% of the total pool was in the soil C pool. The high-latitude zone was estimated to be a C sink increasing by 0.48 $\pm$ 0.2 Pg yr^{-1}. This C sink size is larger than the C sink in mid-latitude forests (0.26 $\pm$ 0.1 Pg yr^{-1}) and the low-latitude forests (in which a relatively large net C source of 1.6 $\pm$ 0.4 Pg $yr^{-1}$was reported). Consequently, the estimated net C flux from the world's forests is a source of 0.9 $\pm$ 0.5 Pg yr^{-1}, or about 16% of the amount produced by burning fossil fuels and cement manufacture. The imbalance between emissions (including emissions from fossil fuel and cement production, and from changes in tropical land use) and sinks (including increases in storage in atmosphere, ocean uptake, and uptake by Northern Hemisphere forest growth) of 1.3 ± 1.5 Pg yr^{-1} is often referred to as the "missing sink", or that amount "needed" to balance the C budget. Substitution of the net C sink for high and mid-latitude forests into the global C budget result in a new difference or "imbalance" of 1.1 ± 1.4 Pg yr^{-1}.

Goodale et al. (2002) summarized forest sector C budgets for Canada, the United States, Europe, Russia, and China and concluded a general agreement that terrestrial systems in the Northern Hemisphere provide a significant sink for atmospheric CO_2; however, estimates of the magnitude and distribution of this sink vary greatly. Together, these suggest that northern forests and woodlands provided a total sink of 0.6–0.7 Pg yr^{-1} of C per year during the early 1990s, consisting of 0.21 Pg yr^{-1} in living biomass, 0.08 Pg yr^{-1} in forest products, 0.15 Pg yr^{-1} in dead wood, and 0.13 Pg yr^{-1} in the forest floor and soil organic matter.

In conceptual understanding of C dynamics, the annual rate of biomass accumulation in live plants and animals, and soil organic matter has been called net ecosystem production (NEP). It represents the aggregated change in all ecosystem biomass pools, and is defined as $NEP = GPP - (R_A + R_H)$, in which GPP is the gross primary production defined as the total amount of C fixed in an ecosystem during the process of photosynthesis; R_A is the total plant respiration and R_H is the respiration of heterotrophic organisms (i.e., C release from dead woody materials and soil through decomposition). The difference between GPP and plant respiration is called net primary production (NPP): $NPP = GPP - R_A$, i.e., it will be higher than NEP by R_H. The proportion of NPP remaining after accounting for losses to herbivory, litterfall, and disturbances is the live biomass accumulation of plants, and it can be either a positive or negative value. This remaining NPP proportion will largely determine the role of a given forest ecosystem in C budget.

Barnes et al. (1998) compiled a diagram of C flux relative to GPP, in which C dynamics in a forest ecosystem can be expressed as a function of years following disturbance (i.e., forest stand age). The diagram showed that the NPP will increase with forest age up to about 60 years and then decline. Without considering disturbance effects, the R_H is mainly determined by the respiration rates from dead woody materials and soils, which is relatively constant over age except a small increase when forest is very young. Because of the lower mean annual temperature in boreal forests, both R_H and NPP could also be lower than the average presented in the diagram. This was confirmed by the research results of temperature and precipitation effect on soil C decomposition rate at equilibrium situation (Yang et al. 2002), where the R_H in boreal forests was generally more than 50% below the average. Based on the mean annual temperature of 2.4 ^{o}C in Boreal West Ecoclimatic province (Kurz et al. 1992), where our study area is located, the R_H could be more than 60% below the average by using Yang's et al. (2002) model, and an even lower R_H value could be expected if the Century model (Parton et al. 1987), which is a well-known soil C dynamics model, is used. Yang's et al. (2002) model predicted similar patterns of temperature effect on R_H and NPP effect across the vegetation types ranging from dry tundra to tropical rain forest, with a slight higher relative value of the NPP than R_H.

Since the GPP can be lower than the R_H when forest is young, a negative NEP can be expected thus the forest ecosystem functions as a C source. Nevertheless, the C balance of forest ecosystem is dynamic and can rapidly become positive following disturbances because of the increasing GPP with forest age. Consequently, whether a given forest ecosystem functions as a C sink or source is largely determined by the sizes and intensities of disturbances such as fire, insect, and harvest (Kurz and Apps 1999; Li and Apps 2003). Furthermore, effects of different disturbances are mainly reflected in the amount of aboveground biomass, the measurement or estimate of aboveground net primary productivity (ANPP) becomes important (Li and Apps 2003), which might be one of the reasons that only the size of living biomass C pool is required by international reporting.

In practice, the frequently-used approach of measuring biomass is based on relationships between the dimensions of forest trees and their weight – allometric biomass equations (Miao and Li 2006). Most of these equations are used to predict the aboveground biomass and productivity of forest ecosystems due to the considerable difficulty in collecting tree roots from the soil.

In current case study, we used operational estimates of merchantable volume for different types of forests (commonly used in timber supply analysis and harvest planning) and converted these estimates to include non-merchantable (such as bark, tops and branches) and belowground volumes. We also used operational forest inventory data that have a fine spatial resolution of 1 ha. Since this resolution is much finer than the 100 km^2 minimum spatial resolution of the national forest inventory data used in the analysis of Goodale et al. (2002), and the biomass C dynamics in each forest stand is tracked in a spatially explicit way in the simulations, we expect this case study could provide a more realistic estimate of C stock dynamics in the study area.

Because of the inclusion of fire disturbance effect (Figure 8F), our results (Figure 8G) reflected the changes in total C sink size per hectare in average. Consequently, our results on the living biomass C stock dynamics in the FALC suggest that the area can generally function as a C sink as long as the fire cycle can be maintained at longer than 100 years.

5.2. Usefulness of current research results

Our research results can be useful in partially meeting Canada's international reporting requirements such as UNFCCC and KP. For example, changing land-use and converting other types of land to forest land can contribute significantly to reach the goal set by the KP; however, suitable and available land that could be transformed to forest land is limited in a forested country like Canada, as indicated in the UN's Global Forest Resource Assessment 2005, where the extent of Canada's forest lands did not change from 1990 to 2005 (Global Forest Resource Assessment 2005). Therefore, the Article 3.4 of the KP, i.e., including managed forests in a nation's GHG accounting, seems a possible option for Canada to increase the C stock in forest lands. However, risks and benefits of adopting Article 3.4 need to be assessed before making the final decision. Consequently, the understanding of C stock changes in managed forests in comparison to unmanaged forests appears essential in determining whether Canada should include it as part of the nation's GHG accounting under the KP.

For the purpose of international reporting, if managed forests are not included in the national C accounting system, the C stock in Canadian boreal forests needs to be estimated and reported using methods approved by IPCC (Penman et al. 2003). If managed forests are included in the national C accounting system, the C stock needs to be evaluated under possible management options such as harvest and burn. Furthermore, it is expected that the forest management policy of emulating natural fire patterns would become increasingly popular in Canada to achieve the goals of sustainable forest resource development and biodiversity conservation. Therefore, assessment of this forest resource management effect on C stock changes should also be incorporated into the analysis. Our current research is closely related to forest management practices in west-central Canada using operational forest inventory data and harvest planning software package. This case study suggests that under a wide range of combinations of forest fire and harvest regimes, this area could still produce a C sink size that is large enough to offset the C released by fire disturbances. Our results thus provided an operational level case study of how the living biomass C stock at the landscape scale could be influenced by the inclusion of various forest management options. The method described in this report can be applied to other forest landscapes of Canada and provide more realistic estimates of C stock across space and time.

5.3. Lessons learned from modeling investigation

There are two main sources that contribute to the diverse results on the role of forests in global C budget reported in the literature: data and methods. Forest inventory data can be at different spatial resolutions. National forest inventory data are generally at a coarse-resolution, while operational forest inventory data have a fine-resolution. C stock estimated by using finer resolution could potentially be more realistic and accurate than that from coarser resolution data. The C stock estimate could be expected to be more reliable when it is obtained from spatially explicit simulations than that of non-spatial simulations, with the cost of a larger computing power being required.

There are different opinions on the role of fire suppression on fire regimes. Construction of time-since-last-fire maps is a standard method in fire history study, and the results often suggest a fire cycle change at some point(s) in history. For example, the historical fire cycles changed around 1900 in Alberta and around 1945 in Saskatchewan. The changes in fire regimes often suggested an increased length of fire cycle, thus, less mean annual area burned should be observed. However, this has not always been clearly evidenced in observations; consequently, many managers and researchers thought the fire frequencies in various regions might not have changed significantly. We did not address this controversy in this report. Instead, we use a range of fire cycles to indicate the possible consequences of various fire protection levels interacting with landscape conditions, select area-weighted mean forest age as an index of forest conditions, and explore how the forest biomass C stock estimate could be associated with the area-weighted mean forest age.

In our research we tried not to make any fundamental assumptions such as the negative exponential forest age distribution before the process-based spatially explicit simulations were carried out. Making these assumptions in research would greatly simplify the simulations; however, at risk is the possible biasing of C stock estimates without reasonable explanation. Research results reported in the literature have revealed that these assumptions might not always be supported by empirical observations (Li and Barclay 2001, 2003). In some cases, simulation results could support and confirm some observations that are not expressed as assumptions build into the models. One example is that the simulated forest productivity and C stock decrease with increasing mean forest age (Figure 6D), which is consistent with recent observations of age-related forest production decline phenomena (Gower et al. 1996; Ryan et al. 1997; Gower and McMurtrie 1999; Binkley et al. 2002; Bereger et al. 2004). Our next step is to simplify the complex by identifying the details that can be eliminated to speed up the simulations without causing significant differences in the results.

Simulated forest C stock dynamics could be greatly influenced by the algorithms of fire regime effects that are implemented. Despite the unchanged simulated fire regime descriptors, the algorithms that implicitly assume a strong age-dependent fire probability function could result in lower estimates of biomass C storage. The details of this issue will be discussed elsewhere; nevertheless, it is the age distribution of the burned forest that will determine the estimate of C released by fire and the remaining C stocks on ground.

Our results suggested that living forest biomass C stock might not be able to be sustained at a higher level through complete protection from any natural and anthropogenic disturbances, due to the natural birth and death process of trees and other vegetation. This is consistent with the conceptual understanding of net primary production (NPP) dynamics with forest age (Barnes et al. 1998). The results may also add some reality to the simplified assumption that younger forests have lower C stocks and older forest have higher C stocks. However, the complexity also allowed forest managers opportunities of enhancing forest C sink sizes through management options. For example, our simulation results suggested that by keeping mean forest age within a certain range (Figure 6D and 9D), the C sequestered could have a higher probability of offsetting C release from fires.

6. Conclusions and Recommendations

6.1. Conclusions obtained from this case study

From our case study results, we notice that forest productivity and C sink size cannot always be sustained without disturbances, which is consistent with the traditional understanding of boreal forests destroyed by fire and created by it. These are supported by the fact that over-mature forests tend to have a very small MAI and net C sequestration capacity, but fuel loads increase to a level that is more prone to fire. Also, native fire-dependent tree species cannot be sustained without fire to open their cones. Therefore, complete protection from any disturbance may not be the best strategy for enhanced C stock in forest lands, and existence of disturbances might not necessarily be bad for C dynamics.

Second, forest age distribution and its dynamics are important in determining forest growth rate such as MAI, and thus the C sequestration, in which a certain range of mean forest age (Figure 6D and 9D) will result in values higher than the default and average values recommended by IPCC. Therefore, keeping forests in the age classes with a high MAI can serve as a target for forest managers. Furthermore, emulating natural fire patterns in harvest planning can increase MAI in young and mid-aged forests, and the C sequestration can increase significantly from 0.46 t/ha, the default value for boreal forests, to up to 1.2 t/ha (Figure 9D). The FALC forests in most cases will function as C sinks except when a fire cycle becomes very short (< 100 years), hence the area has the potential to contribute positively to the C sink. The only uncertainty is the future fire regime in the area, and we need to explore whether this would change significantly under future climate and forest conditions, where a threshold fire cycle could switch the role of forests from a C sink to a C source.

Third, if the C transferred from living biomass to forest products pool is considered in the C accounting process, the FALC forests will even tend to function as a C sink (Figure 8I).

6.2. Recommendations for future research from this case study

- To carry out more case studies for dissimilar geographic regions with different forest compositions and landscape conditions. In these case studies, C accounting systems should involve the wood/timber supply research community and apply spatially explicit and mechanistic simulation approaches using operational forest inventory data wherever they are available. In this way, realistic and reliable forest C stock dynamics can be obtained.
- To develop a unified simulation framework that facilitates the inclusion of various major ecosystem components and their interactions for same or similar land bases. Our current research included fire and harvest regimes, and future studies need to consider other important disturbance regimes as well, such as MPB that has been recognized as a major threat under current climate conditions.
- To perform a comprehensive economic analysis before implementing research results in forest management practice. This analysis should contain timber and non-timber values, wildlife habitat, biodiversity, costs and benefits from different management options including C credit calculation and the associated values. A decision-making package can then be generated for forest managers and stakeholders.
- To include C dynamics in dead woody materials and soils despite the reality that they are not currently required for international reporting purpose. Inclusion of the measurements of C dynamics in dead woody materials and soils could enable forest managers and researchers to address the question of whether the C release through decomposition and respiration would be big enough to offset C sequestration in living biomass with the intent that the role of forests in global C budget could be redefined.

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References

Alberta Sustainable Resource Development. 2004. 2004 Annual Report: Forest Health in Alberta. Department of Sustainable Resource Development, Public Lands and Forests Division, Forest Management Branch, Forest Health Section. Edmonton, Alberta. 54 p. <http://www3.gov.ab.ca/srd/forests/health/p_reports.html> Accessed on Feb. 15, 2006.

Albinet, G., Searby, G. and Stauffer, D. 1986. Fire propagation in a 2-D random medium. J. Physique 47: 1-7.

Albini, F. A. 1976. Estimating wildfire behavior and effects. U. S. Dep. Agric., For. Serv., Interm. For. Range Exp. Stn., Ogden, UT, GTR-30.

Albini, F. A. 1979. Spot fire distance from burning trees – a predictive model. U.S. Dep. Agric., For. Serv., Rocky Mt. Res. Stn. Ogden, UT. Res. Pap. INT-56,

Albini, F. A. 1983. Potential spotting distance from wind-driven surface fires. U.S. Dep. Agric., For. Serv., Rocky Mt. Res. Stn. Ogden, UT. Res. Pap. INT-309.

Alexander, M. E. 1985. Estimating the length-to-breadth ratio of elliptical forest fire patterns. P. 287-304 in Proc. 8th Conf. Fire and For. Meteorol., April 29-May 2, 1985, Detroit, Michigan. Soc. Am. For., Bethesda, Maryland, MD.

Anderson D. G., Catchpole, E. A., DeMestre, N. J., and Parkes, T. 1982. Modeling the spread of grass fires. J. Aust. Math. Soc. (Ser. B.) 23: 451-466.

Anderson, K., Powell, S., and Etches, M. 1998. A modified suppression response decision support system for Wood Buffalo National Park. P. 32-37 in Proc. 2nd Symp. Fire For. Meteorol., Jan. 11-16, 1998, Phoenix, Arizona.

Baker, W. L. 1995. Longterm response of disturbance landscapes to human intervention and global change. Landsc. Ecol. 10: 143-159.

Bhatti, J.S., van Kooten, G.C., Apps, M.J., Laird, L.D., Campbell, I.D., Campbell, C., Turetsky, M.R., Yu, Z., and Banfield, E. 2003. Carbon balance and climate change in boreal forests. Chapter 20. Pages 799-855 in Burton, Pl.j., Messier, C., Smith, D.W., and Adamowicz, W.L., eds. Towards Sustainable Management of the Boreal Forest. NRC Research Press, Ottawa, Ontario, Canada.

BC Ministry of Forests and Range. 1996. Forest Practices Code: Timber supply analysis. < http://www.for.gov.bc.ca/tasb/legsregs/fpc/pubs/timbersupply/tsa-toc.htm> Accessed on Feb. 15, 2006.

BC Ministry of Forests. 2003. Timber supply and the mountain pine beetle infestation in British Columbia. < http://www.for.gov.bc.ca/hts/pubs/beetledoc_oct29LO.pdf> Accessed on Feb. 15, 2006.

Beck, J. A. and Trevitt, A. C. F. 1989. Forecasting diurnal variations in meteorological parameters for predicting fire behaviour. Can. J. For. Res. 19: 791-797.

Berger, U., Hildenbrandt, H., and Grimm, V. 2004. Age-related decline in forest production: modelling the effects of growth limitation, neighbourhood competition and self-thinning. J. Ecol. 92: 846-853.

Bessie, W. C. and Johnson, E. A. 1995. The relative importance of fuels and weather on fire behavior in subalpine forests. Ecology 76: 747-762.

Binkley, D., Stape, J.L., Ryan, M.G., Barnard, H.R., and Fownes, J. 2002. Age-related decline in forest ecosystem growth: an individual-tree, stand-structure hypothesis. Ecosystems 5: 58-67.

Bradbury, R. H., Green, D. G., and Snoad, N. 2000. Are ecosystems complex systems? Pages 339-365 in Bossomaier, R. J. and Green, D. G. eds. Complex systems. Cambridge Univ. Press. Cambridge, UK.

Brown, S. 1996. The world's forest resources. Unasylva 43: 3-10.

Brown, S. 1997. Forests and climate change: role of forest lands as carbon sinks. Pages 117-129 in Proceedings of the 11th World Forestry Congress, held in Antalya, Turkey, October 1997.

Brown, S., Sathaye, J., Cannell, M., Kauppi, P.E. 1996. Management of forests for mitigation of greenhouse gas emissions. Pages 773-798 in Watson, R.T., Zinyowera, M.C., and Moss, R.H. (eds.) Climate change 1995: impacts, adaptations and mitigation of climate change: scientific analyses. Contribution of working group II to the second assessment report of the ntergovernmental Panel on Climate Change. Cambridge Univ. Press, Cambridge.

Bruntland, G. (ed.) 1987. Our Common Future: World Commission on Environment and Development. Oxford Univ. Press. 398 pages.

Barnes, B. V., Zak, D. R., Denton, S. R., and Spurr, S. H.: 1998, Forest Ecology. 4th Edition. John Wiley & Sons, Inc. New York, NY.

Canadian Council of Forest Ministers. 1997. Compendium of Canadian forestry statistics 1996. Can. Counc. For. Minist., Ottawa, Ont.

Cardy, J. L. and Grassberger, P. 1985. Epidemic models and percolation. J. Phys. A. Math. Gen. 18: L267-L271.

Ciais, P., Tans, P. P., Trolier, M., White, J. W. C., and Francey, R. J. 1995. A large northern hemispheric terrestrial CO_2 sink indicated by the 13C/12C ratio of atmospheric CO_2. Science 269: 1098-1102.

Costanza, R. and Jørgensen, S.E. 2002. Understanding and solving environmental problems in the 21st century: toward a new, integrated hard problem science. Elsevier Science Ltd. The Boulevard, Langford Lane, Kidington, Oxford, UK. 324 pages.

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. Nat. Resour. Can., Can. For. Serv., North. For. Cent., Edmonton, Alta., and Alta. Environ. Prot., Land For. Serv., Edmonton Alta. Can.-Alta. Partnership Agreement For. 137.

Dixon, R. K., Brown, S., Houghton, R. A., Solomon, A. M., Trexler, M. C., and Wisniewski, J. 1994. Carbon pool and flux of global forest ecosystems. Science 263: 185-190.

Finney, M. A. 1994. Modeling the spread and behavior of prescribed natural fires. Pages 138-143 in Proc. 12th Conf. Fire For. Meteorol., October 26-28, 1993, Jekyll Island, Georgia. Soc. Am. For., Bethesda, Maryland. MD.

Finney, M. A. 1998. FARSITE: fire area simulator, model development and evaluation. U.S. Dep. Agric., For. Serv. Rocky Mt. Res. Stn., Ogden, UT. Res. Pap. RMRS-4.

Finney, M. A. 1999. Mechanistic modeling of landscape fire patterns. P. 186-209 in Spatial modeling of forest landscape change: approaches and applications. D. J. Mladenoff, and W. L. Baker, eds. Cambridge Univ. Press, Cambridge, UK.

Forestry Canada Fire Danger Group. 1992. Development and structure of the Canadian Forest Fire Behaviour Prediction Systems. For. Can., Sci. Sustainable Dev. Dir. Ottawa, Ont. Inf. Rep. ST-X-3.

Fosberg, M. 1992. International Boreal Forest Research Association Stand Replacement Fire Working Group. IFFN No. 7 – August 1992, pages 6-8. < http://www.fire.uni-freiburg.de/iffn/country/rus/rus_8.htm> Accessed on Feb. 15, 2006.

Gardner, R. H., Hargrove, W. W., Turner, M. G., and Romme, W. H. 1996. Climate change, disturbances and landscape dynamics. P 289-307 in Walker, B. H. and Steffen, W. L. eds. Global change and terrestrial ecosystems. Int. Geosphere-Biosphere Program Book Ser. 2. Cambridge Univ. Press, Cambridge, UK.

Global Forest Resource Assessment 2005. <http://www.fao.org/forestry.fra2005> Accessed on Feb. 15, 2006.

Goodale, C.L., Apps, M.J., Birdsey, R.A., Field, C.B., Heath, L.S., Houghton, R.A., Jenkins, J.C., Kohlmaier, G.H., Kurz, W., Liu, S., Nabuurs, G.J., Nilsson, S., and Shvidenko, A.Z. 2002. Foret carbon sinks in the northern hemisphere. Ecological Applications 12: 891-899.

Gower ST, McMurtrie RE. 1999. An analysis of the age-related decline in aboveground net primary production; potential causes and stand-to-global scale implications. Santa Barbara (CA): NCEAS, University of Santa Barbara.

Gower ST, McMurtrie RE, Murty D. 1996. Aboveground net primary production decline with stand age: potential causes. Trends Ecol. Evol. 11:378–82.

Granström, A. 1993. Spatial and temporal variation in lighning ignitions in Sweden. J. Veg. Sci. 4: 737-744.

Grassberger, P. 1983. On the critical behavior of the general epidemic process and dynamical percolation. Math. Biosci. 62: 157-172.

Grimm, V. and Failsback, S.F. 2005. Individual-based Modeling and Ecology. Princeton Univ. Press, Princeton, New Jersey USA.

Hargrove, W. W., Gardner, R. H., Turner, M. G., Romme, W. H., and Despain, D. G. 2000. Simulating fire patterns in heterogeneous landscapes. Ecol. Model. 135: 243-263.

Heinselman, M. L. 1973. Fire in the virgin forest of the Boundary Waters Canoe Area, Minnesota. Quat. Res. 3: 329-382.

Holling, C. S., Peterson, G., Marples, P., Sendzimir, J., Redford, K., Gunderson, L., and Lambert, D. 1996. Self-organization in ecosystems: lumpy geometries, periodicities, and morphologies. Pages 346-384 in Walker, B. H. and Steffen, W. L. eds. Global change and terrestrial ecosystems, Cambridge Univ. Press, Cambridge, UK.

Houghton, R. A.: 1996, Terrestrial sources and sinks of carbon inferred from terrestrial data. Tellus Ser. B Chem. Phys. Meteorol. 48: 430-433.

IPCC (Intergovernmental Panel on Climate Change. 2001. Climate change 2001: the scientific basis. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate change. Edited by Houghton, J.T., Ding, Y., Griggs, K.D., Noguer, M., van der Linden, P.J., and Xiao, D. Cambridge Univ. Press, Cambridge, U.K.

Johnson, E.A. and Gutsell, S.L. 1994. Fire frequency models, methods and interpretations. Adv. Ecol. Res. 25: 239-287.

Kauppi, P. E., Mielikainen, K., and Kuusela, K.: 1992, Biomass and carbon budget of European forests, 1971 to 1990. Science 256: 70-74.

Keane, R. E., Morgan, P., and Running, S. W. 1996. FIRE-BGC: a mechanistic ecological process model for simulating fire succession on coniferous forest landscapes of the northern Rocky Mountain. U.S. Dep. Agric., For. Serv. Intermt. Res. Stn., Ogden, UT. Res. Pap. INT- 434.

Keane, R.E., K. Ryan & S.W. Running, 1995. Simulating the effects of fire and climate change on northern Rocky Mountain landscapes using the ecological process model FIRE-BGC. USDA For. Serv. Gen. Tech. Rep. RM-262. 39-47.

Keane, R. E., Ryan, K. C., and Running, S. W. 1996b. Simulating effects of fire on northern Rocky Mountain landscape with the ecological process model FIRE-BGC. Tree Phys. 16: 319-331.

Keane, R.E., Cary, G.J., Davies, I.D., Flannigan, M.D., Gardner, R.H., Lavorel, S., Lenihang, J.M., Li, C. and Ruppi, T.S. 2004. A classification of landscape fire succession models: spatial simulations of fire and vegetation dynamics. Ecol. Model. 179: 3-27.

Kimmins, J.H. 2004. Emulating natural forest disturbance: what does this mean? In Perera, A.H., Buse, L.J., and Weber, M.G. (Eds). Emulating Natural Forest Landscape Disturbances: Concepts and Applications. Pages 8-28.

Knight, I. and Coleman, J. 1993. A fire perimeter expansion algorithm based on Huygen's wavelet propagation. Int. J. Wildland Fire 3: 73-84.

Kourtz, P., Nozaki, S., and O'Regan, W. 1977. Forest fires in the computer: a model to predict the perimeter location of a forest fire. Fish. Environ. Can., Ottawa, Ont. For. Fire Res. Inst. Inf. Rep. FF-X-65.

Kurz, W.A. and Apps, M.J. 1999. A 70-year retrospective analysis of carbon fluxes in the Canadian forest sector. Ecol. Appl. 9: 526-547.

Kurz, W. A., Apps, M. J., Webb, T. M., and McNamee, P. J.: 1992, The carbon budget of the Canadian forest sector: Phase I. For. Can., Northwest Reg., North. For. Cent., Inf. Rep. NOR-X-326.

Landsberg, J.J. and Waring, R.H. 1997. A generalised model of forest productivity using simplified concepts of radiation-use efficiency, carbon balance and partitioning. For. Ecol. Manag., 95, 209-228.

Latham, D. J. and Rothermel, R. C. 1993. Probability of fire-stopping precipitation events. U.S. Dep. Agric., For. Serv., Intermt. Res. Stn., Ogden, UT. Res. Note INT-410.

Li, C. 2000a. Reconstruction of natural fire regimes through ecological modelling. Ecol. Model. 134, 129-144.

Li, C. 2000b. Fire regimes and their simulation with reference to Ontario. P. 115-140 in Perera, A., Euler, D., and Thompson, I. eds. Ecology of a managed terrestrial landscape: patterns and processes of forest landscapes in Ontario. Univ. British Columbia Press. Vancouver, B.C.

Li, C. 2000c. Modeling the influence of fire ignition source patterns on fire regimes of west-central Alberta. In: Proc. 4th Intl. Conf. Integrating GIS and Environmental Modeling (GIS/EM4): Problems, Prospects and Research Needs. Banff, Alberta, Canada, September 2-8, 2000. (To be on CD-ROM) Available now on web: <http://www.Colorado.edu/research/cires/banff/upload/92> Latest access date: July 9, 2001.

Li, C. 2002. Estimation of fire frequency and fire cycle: a computational perspective. Ecol. Model. 154: 103-120.

Li, C. 2004. Simulating forest fire regimes in the foothills of the Canadian Rocky Mountains. Pages 98 – 111 in Perera, A.H., Buse, L.J. and Weber, M.G. eds. Emulating Natural Forest Landscape Disturbances: Concepts and Applications. Columbia Univ. Press. New York, NY.

Li, C. and Apps, M. J. 1995. Disturbance impact on forest temporal dynamics. Water Air Soil Pollut. 82: 429-436.

Li, C. and Apps, M. J. 1996. Effects of contagious disturbance on forest temporal dynamics. Ecol. Model. 87: 143-151.

Li, C. and Apps, M.J. 2003. Fire regimes and the carbon dynamics of boreal forest ecosystems. Pages 107-118 in: Shaw, C.H. and Apps, M.J. eds. Proceedings of the International conference of "The role of boreal forest and forestry in the global carbon budget" in Edmonton, Alberta, May 8-12, 2000.

Li, C. and Barclay, H.J. 2001. Fire disturbance patterns and forest age structure. Nat. Resour. Model. 14: 495-521.

Li, C. and Barclay, H. 2004. Simulation of Interactions among fire, mountain pine beetle and lodgepole pine forest. Pages 257-266 in Shore, T.L., Brooks, J.E., and Stone, J.E. eds. Mountain Pine Beetle Symposium: Challenges and Solutions. Can. For. Serv., Inf. Rep. BC-X-399. Victoria, B.C.

Li, C., Corns, I. G. W., and Yang, R. C. 1999. Fire frequency and size distribution under natural conditions: a new hypothesis. Landsc. Ecol. 14: 533-542.

Li, C., Flannigan, M.D., and Corns, I.G.W. 2000. Influence of potential climate change on forest landscape dynamics of west-central Alberta. Can. J. For. Res. 30: 1905-1912.

Li, C., Barclay, H., Liu, J. and Campbell, D. 2005. Simulation of historical and current fire regimes I central Saskatchewan. For. Ecol. Manag. 208: 319-329.

Li, C., Barclay, H.J., Hawkes, B.C., and Taylor, S.W. 2005. Lodgepole pine forest age class dynamics and susceptibility to mountain pine beetle attack. Ecol. Complex. 2: 232-239.

Li, C., Ter-Mikaelian, M., and Perera, A. 1997. Temporal fire disturbance patterns on a forest landscape. Ecol. Model. 99: 137-150.

McGarigal, K., Marks, B.J., 1995. FRAGSTATS: spatial pattern analysis program for quantifying landscape structure. Gen. Tech. Rep. PNW-351. USDA For. Serv.

MacKay, G. and Jan, N. 1984. Forest fires as critical phenomena. J. Phys. A. Math. Gen. 17: 163-169.

Malamud, B. D., Morein, G., and Turcoptte, D. L. 1998. Forest fires: an example of self-organized critical behavior. Science 281: 1840-1842.

Merrill, D. F. and Alexander, M. E. (Ed.) 1987. Glossary of forest fire management terms. 4th Ed. Natl. Res. Counc. Can., Can. Comm. For. Fire Manag., Ottawa, Ont. Publ. NRCC No. 26516.

Miao, Z. and Li, C. 2006. Comparison of biomass equations for west-central Canada's boreal forests. (Submitted, in review)

Minnich, R. A. 1983. Fire mosaics in Southern California and Northern Beja California. Science 219: 1287-1294.

Von Mirbach, M. 2000. Carbon budget accounting at the forest management unit level: an overview of issues and methods. Canada's Model Forest Program, Can. For. Serv., Nat. Resour. Can., Ottawa, ON. < http://dsp-psd.pwgsc.gc.ca/Collection/Fo42-312-2000E.pdf> Accessed on Feb. 15, 2006.

Von Niessen, W. and Blumen, A. 1986. Dynamics of forest fires as a directed percolation model. J. Phys. A. Math. Gen. 19: L289-L293.

Von Niessen, W. and Blumen, A. 1988. Dynamic simulation of forest fires. Can. J. For. Res. 18: 805-812.

Ohtsuki, T. and Keyes, T. 1986. Biased percolation: forest fires with wind. J. Phys. A. Math. Gen. 19: L281-L287.

O'Neill, R. V., Gardner, R. H., Turner, M. G. and Roome, W. H. 1992. Epidemiology theory and disturbance spread on landscapes. Landsc. Ecol. 7: 19-26.

Parton, W.J., Schimel, D.S., Cole, C.V., and Ojima, D.S. 1987. Analysis of factor controlling soil organic levels in Great Plains grasslands. Soil Sci. Soc. Am. J. 51: 1173-1179.

Penman, J., Gytarsky, M., Hiraishi, T., Krug, T., Kruger, D., Pipatti, R., Buendia, L., Miwa, K., Ngara, T., Tanabe, K., and Wagner, F. (eds.) 2003. IPCC Good Practice Guidance for Land Use, Land-Use Change and Forestry. IPCC National Greenhouse Gas Inventories Programme.

Peterson, G. D. 1994. Spatial modelling of fire dynamics in the Manitoba boreal forest. M.Sc. thesis, Univ. Florida, Gainsville, FL.

Phipps, M. J. 1989. Dynamical behavior of cellular automata under the constraint of neighborhood coherence. Geogr. Anal. 21: 197-215.

Phipps, M. J. 1992. From local to global: the lesson of cellular automata. P. 165-187 in DeAngelis, D. L. and Gross, L. J. eds. Individual based models and approaches in ecology: populations, communities and ecosystems. Routeledge, Chapman and Hall, New York, N.Y.

Richards, G. D. 1994. The properties of elliptical wildfire growth for time dependent fuel and meteorological conditions. Combust. Sci. Tech. 95: 357-383.

Richards, G. D. and Bryce, R. W. 1995. A computer algorithm for simulating the spread of wildland fire perimeters for heterogeneous fuel and meteorological conditions. Int. J. Wildland Fire 5: 73-80.

Riggan, P. J., Goods, S., Jacks, P. M. and Lockwood, R. N. 1988. Interaction of fire and community development in chaparral of southern California. Ecol. Monogr. 58: 155-176.

Rothermel, R. C. 1991. Predicting behavior and size of crown fires in the northern Rocky Mountains. U.S. Dep. Agric., For. Serv., Intermt. Res. Stn., Ogden, UT. Res. Pap. INT-438.

Ryan, M.G., Binkley, D., Fownes, J.H. 1997. Age-related decline in forest productivity: pattern and process. Adv. Ecol. Res. 27:213–62.

Saskatchewan Environment. 1999. Fort A La Corne integrated forest land use plan – background information. < http://www.se.gov.sk.ca/forests/landuse/fort/toc.htm> Accessed on Feb. 15, 2006.

Sauchyn, D. J. and Beaudoin, A. B. 1998. Recent environmental change in the southwestern Canadian Plains. Can. Geogr. 42: 337-353.

Sedjo, R.A. 1992. Temperate forest ecosystems in the global carbon cycle. Ambio 21: 274-277.

Simard, A. and Young, A. 1978. AIRPRO: an air tanker productivity computer simulation model – the equations (documentation). Fish. Environ., Can., Ottawa, Ont., For. Fire Res. Inst. Inf. Rep. FF-X-66.

Spatial Planning Systems. 2004. < http://www.spatial.ca/index.html> Accessed on Feb. 15, 2006.

Stocks, B.J., Goldammer, J.G., Fosbert, M., Conard, S., and Valendik, E. 1996. International Cooperation in Boreal Forest Fire Research: The IBFRA Stand Replacement Fire Working Group. Int. For. Fire News No. 15, 54-58. <http://www.fire.uni-freiburg.de/programmes/other/Stocks.html> Accessed on Feb. 15, 2006.

Turner, M. G., Hargrove, W. W., Gardner, R. H. and Romme, W. H. 1994. Effects of fire on landscape heterogeneity in Yellowstone National Park, Wyoming. J. Veg. Sci. 5: 731-742.

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. Fire behavior in northern conifer forests and shrublands. P. 65-80 in Wein, R. W. and MacLean, D. A. eds. The role of fire in northern circumpolar ecosystems. John Wiley & Sons. New York, N.Y.

Van Wagner, C. E. 1987. Development and structure of the Canadian Forest Fire Weather Index System. Ottawa, Ont. Can. For. Serv. For. Tech. Rep. No. 35.

Walters, K.R. and Cogswell, A. 2002. Spatial forest planning: where did all the wood go? Forest Technology Group, Remsoft Inc. < http://www.remsoft.com/docs/library/spatial.forest.planning.pdf> Accessed on Feb. 15, 2006.

Watson, R., Noble, I. R., Bolin, B., Ravindranath, N.H., Verado, D. J., and Dokken, D. J. 2000. Land Use, Land-Use Change and Forestry. Special Report of the IPCC, Cambridge Univ. Press. 377 p.

Weber, M. G. and Flannigan, M. D. 1997. Canadian boreal forest ecosystem structure and function in a changing climate: impact on fire regimes. Environ. Rev. 5: 145-166.

Wiitala, M. and Carlton, D. 1994. Assessing long-range fire movement risk in wilderness fire management. P. 187-194 in Proc. 12th Conf. Fire For. Meteorol., Oct. 26-28, 1993, Jekyll Island, Georgia. Soc. Am. For., Bethesda, MD.

Wolfe, S.A. and Ponomarenko, D.S. 2001. Potential sediment transport in the prairie provinces from principal climate station data. CCAF Science Project #S00-15-09. Unpublished Report Submitted to Environment Canada.

Wolfram, S. 1985. Some recent results and questions about cellular automata. P. 153-167 in Demongeot, J., Goles, E., and Tchuente, M. eds. Dynamical systems and cellular automata. Academic Press, London, UK.

Wu, Y., Sklar, F.H., Gopu, K., and Rutchey, K. 1996. Fire Simulations in the Everglades Landscape Using Parallel Programming. Ecol. Model. 93:113-124.

Yang, X, Wang, M., Huang, Y. and Wang, Y. 2002. A one-compartment model to study soil carbon decomposition rate at equilibrium situation. Ecol. Model. 151:63-73