LIMITED REPORT

Sensitivity of Boreal Forest Landscapes to Climate Change

by

M. Johnston

Environment Branch

SRC Publication No. 11341-7E01

technology is our business

Canada

June, 2001

LIMITED REPORT

Sensitivity of Boreal Forest Landscapes to Climate Change

by

M. Johnston

Environment Branch

Saskatchewan Research Council
15 Innovation Blvd.
Saskatoon, SK S7N 2X8
Phone: 306-933-7432
Fax: 306-933-7817

SRC Publication No. 11341-7E01

June, 2001

TABLE OF CONTENTS

page

LIST OF TABLES

LIST OF FIGURES

INTRODUCTION AND OBJECTIVES

The world's boreal forests are expected to be severely affected by climate change in the next 100 years. For example, the recent Intergovernmental Panel on Climate Change (IPCC) report (Watson et al. 1996) suggests that the boreal forest is expected to experience the greatest impact, because warming is expected to be particularly large at high latitudes, and because boreal forests are more strongly affected by temperature than forests in other latitudinal zones (Kirschbaum and Fischlin 1996). Climate change impacts in these areas will be felt directly, through changes in temperature and precipitation (e.g. Herrington et al. 1997, Singh and Wheaton 1991) and indirectly, through changes in the disturbance regime. Of particular importance to boreal forest ecosystems are changes to disturbances due to fire (Flannigan et al. 1998, 2001; Stocks et al. 1998) and insects (Fleming 2000, Volney and Fleming 2000).

Information on climate change impacts currently available to forest managers is at temporal and spatial scales that are not relevant to planning and management activities (O'Shaughnessy 2001). According to Cash (2000),

> *Global* mean temperature change, while perhaps spurring international action, is irrelevant to *local* emergency relief managers in Bangladesh or farmers in Nebraska (emphasis in the original, p. 242).

The objective of this project was to gather currently available information on climate change impacts to the western Canadian boreal forest and apply these data to three example landscapes across Saskatchewan. We chose landscapes of an extent (*ca.* 100 km^2) consistent with the scale at which forest managers carry out planing and management activities. Particular emphasis was placed on using data currently available and familiar to forest managers, rather than relying on complicated modelling analyses. This allowed us to determine whether this type of analysis could be carried out by forest managers who do not have an extensive modelling background. We also wanted to address the need for a synthesis of information already available, since it is the interactions among the various stressors that will determine the ultimate ecosystem impacts of climate change (Stocks 1987, Hogg and Hurdle 1995, Fleming 2000). The interactions we recognized in this study are shown in Figure 1.

METHODS

The three study areas were chosen to represent a range of vegetation, soil and climatic characteristics. Our original intention was to use study sites from both Saskatchewan and Manitoba but data availability and lack of consistency across provincial boundaries prevented this from occurring. Instead, we chose three areas from across central Saskatchewan (Figure 2). These were (from west to east) Bronson Forest (BF), Montreal Lake (ML) and Hudson Bay (HB). Each is a 10 km x 10 km UTM map sheet, 10,000 ha in area. Table 1 shows the location and ecological characteristics of each study area.

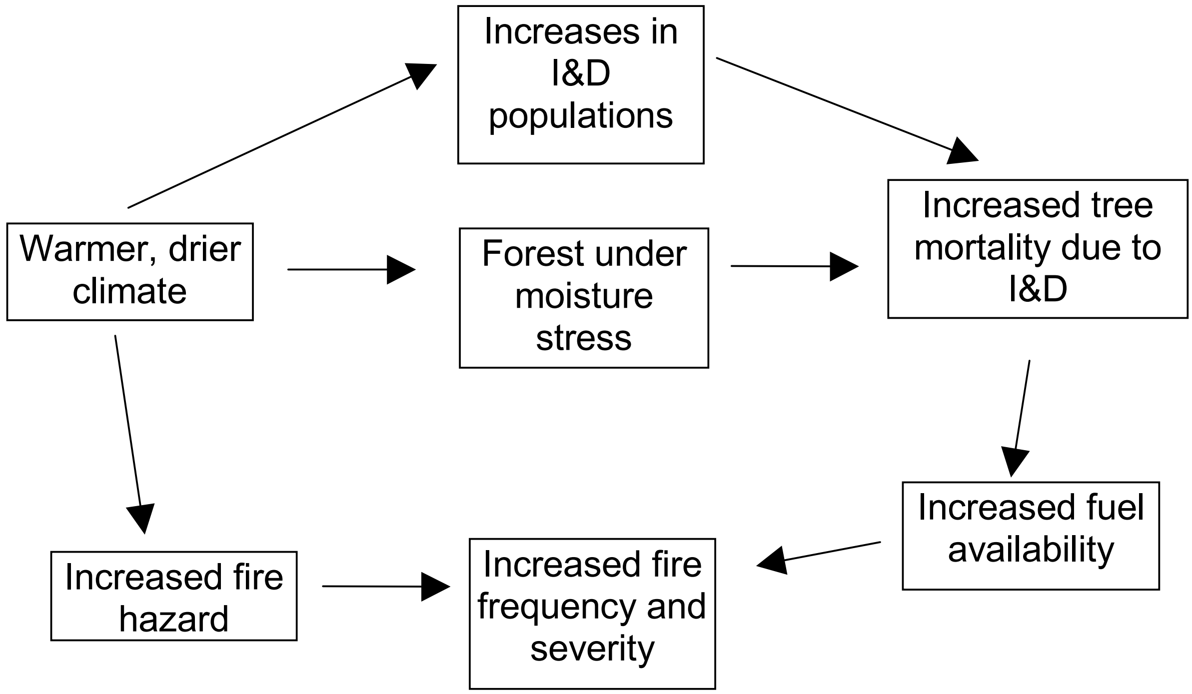

Figure 1 **A conceptual model of interactions among ecosystem components likely to be important under climate change scenarios for the western Canadian boreal forest** (based on Stocks 1987, Hogg and Hurdle 1995, Stocks et al. 1998 and Fleming 2000).

Table 1 **Location and ecological characteristics for three study sites in the southern boreal forest of Saskatchewan.**

	Bronson Forest	Montreal Lake	Hudson Bay
Latitude Longitude	53° 52' 14" N 109° 37' 52" W	54° 46' 29" N 105° 46' 38" W	52° 20' 05" N 103° 05' 31" W
Mean Elevation (masl)	614	433	592
Upland Forest (ha)	6769	6288	7069
Mean Annual Temp (°C) (1961-1990)	0.68	-0.43	0.43
Mean Annual Precipitation (1961-1990)	415.6	509.3	480.7

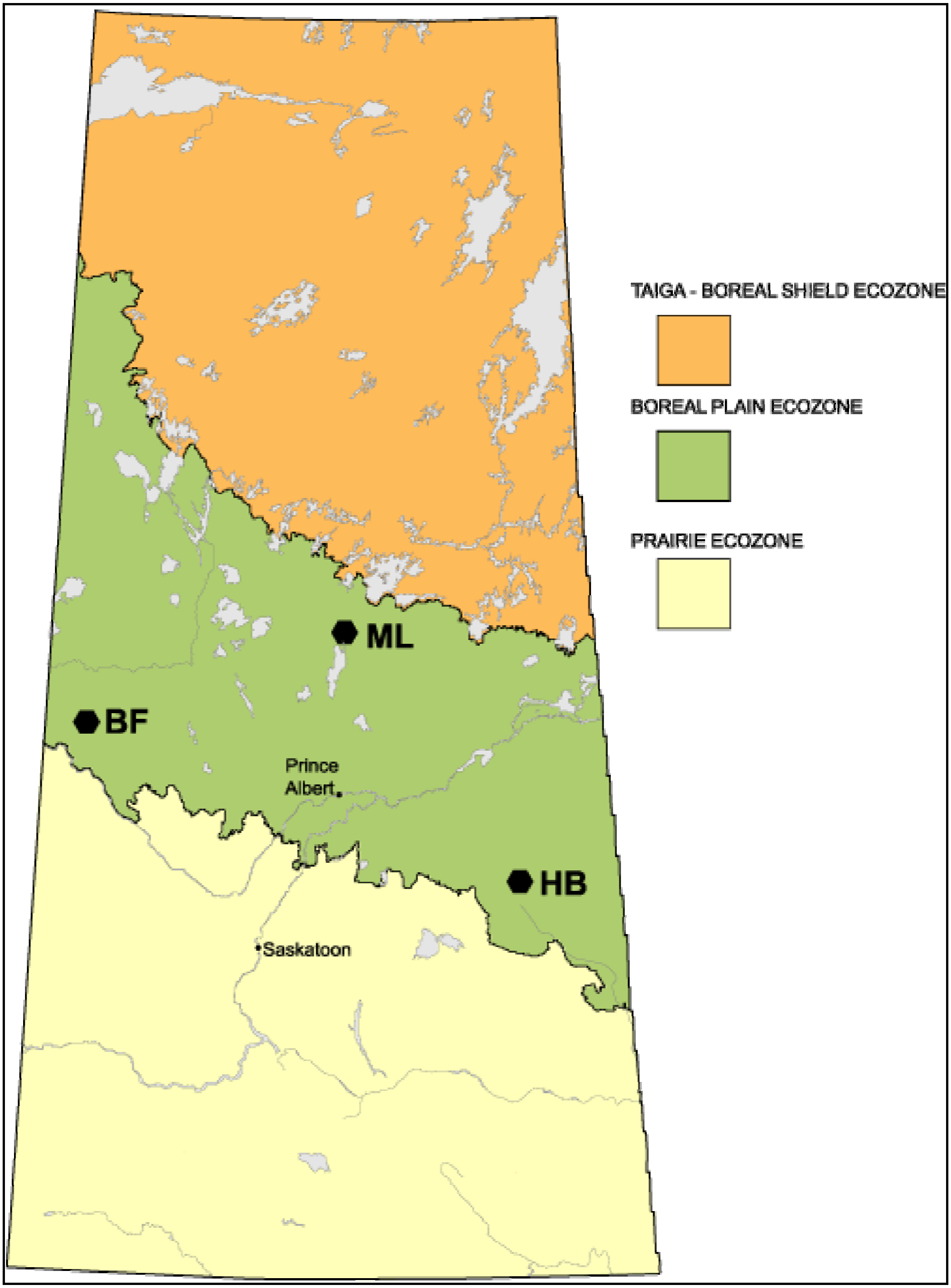

Figure 2 **Location of study areas in central Saskatchewan. Areas are (from west to east) Bronson Forest (BF), Montreal Lake, (ML) and Hudson Bay (HB).**

Data Sources

Data assembled for each area included:

1. **Climate:** The 1961-1990 Climate Normals for the ecodistrict within which each area occurred, taken from the CANSIS database maintained by Research Branch, Agriculture and Agrifood Canada (http://sis.agr.gc.ca/cansis/nsdb/ecostrat/district/climate.html). For the future climate, climate change scenarios were produced by the Canadian Climate Impacts Scenarios research group (http://www.cics.uvic.ca/scenarios/index.cgi). Scenarios were developed from output produced by the Canadian Global Circulation Model, Version 1, greenhouse gas-only runs (CGCM1-GG) for the years 2010-2039 (referred to hereafter as 2020), 2040-2069 (2050) and 2070-2100 (2080). These scenarios represent the maximum projected warming produced by the CGCM.

2. **Vegetation:** Standard provincial forest inventory coverage was used for the Bronson Forest and Hudson Bay map sheets. For the Montreal Lake map sheet more recent forest inventory data was used as collected by the Forest Management Agreement holder, Weyerhaeuser Canada, Inc.

3. **Soils:** Digital soils data were obtained from the Land Resources Unit, Saskatchewan Soil Research Centre, Agriculture and Agri-Food Canada located at the University of Saskatchewan. For the Bronson Forest and Hudson Bay study areas, existing standard soil survey data were used as described by Shields et al. (1991). For the Montreal Lake area, recent soils data collected by the Land Resource Unit were used. For each upland soil polygon, available water-holding capacity (AWC) was estimated from soil texture for the upper 120 cm of the soil profile (Shields et al. 1991).

4. **Forest Fire:** Data on forest fire intensity under current and future climates were provided by the Fire Research Network, Northern Forestry Centre, Canadian Forest Service, Edmonton AB. These data comprised head fire intensity estimates for each fuel type that occurred on each map sheet. Estimates were determined for historical climate data and for two climate scenarios: 2 x current CO_2 levels ($2xCO_2$) and $3xCO_2$. These scenarios were developed from output produced by the Canadian Regional Climate Model analysis for western Canada described by Laprise et al. (1998).

Analysis

Moisture availability

Three climate change impact factors were investigated in detail: soil moisture stress and its impact on potential biomass productivity, changes in potential forest fire intensity, and changes in susceptibility to insect infestations. For the soil moisture stress analysis, a forest water balance model (WATBAL) was used, provided by Dr. Michael Starr, Finnish Forest Research Institute, Vantaa, Finland (available at http://www.metla.fi/hanke/3098/ewat_bal.htm). Inputs to the model include monthly means of precipitation (mm), temperature (°C) and cloudiness (fraction of sky cover). In addition, user-adjustable parameters for the model include latitude (decimal degrees), elevation (masl), aspect (deg.) and slope(deg.), forest canopy coverage (%), long-term mean daily minimum and maximum temperatures for the warmest month (°C), initial amount of snow on the ground at the beginning of the model run (mm), and initial soil moisture content (mm). To represent the current climate the 1961-1990 normal climate data from the CANSIS database was used as described above, and to represent the future climate scenarios the CGCM1 output for 2020, 2050 and 2080 for the gridpoint nearest each study area was used. The model was run under each of the four climate scenarios at five levels of AWC: 50 mm, 100, 150, 200, 250 mm. These are the same values as those used in the CANSIS soils database to characterize soil water holding capacity and are also those used in the soil moisture deficit calculations in the CANSIS ecodistrict normal climate data.

Outputs from the model include actual evapotranspiration (AET), potential evapotranspiration (PET), global radiation, water equivalent of snow cover, snowmelt, soil moisture content and soil water flux (Starr 1999). The ratio of AET to PET (AET/PET) was used as a general index of moisture demand and supply for the study sites under both current and future climates. AET/PET is a simple index of moisture stress relevant to tree species occurrence and growth (Shafer et al. 2001). For the current (1961-1990) climate, temperature and precipitation values were used from the CANSIS database described above, and for the future climate, temperature and precipitation values from the CGCM1-GG model were used as input to WATBAL. The model then provided AET and PET as output for each climate scenario. In addition, the relationship between potential soil moisture deficits and soil water-holding capacity was examined under a drier future climate, and the model output was used to calculate a biomass productivity index (see below). The combination of four climate scenarios, five AWC values and three study areas resulted in a total of 60 water balance analyses.

Productivity

The effect of changes in soil moisture availability and temperature on current and potential future forest productivity were examined. This was accomplished by calculating an index of biomass productivity originally described by Turc and Lecerf (1972). Previous work in Alberta (Williams 1985) and Saskatchewan (Wheaton and Wittrock 1994, Williams and Wheaton 1998) has shown this index to be a reasonable predictor of changes in potential biomass productivity under current and future climate change scenarios. The index accounts for the effects of temperature, solar radiation and soil moisture availability on potential biomass production. Temperature and radiation are combined to form the "heliothermic factor" as shown in equation 1:

$$Ft \times Fh = Ht, \qquad (1)$$

where Ft is the "thermal factor" based on mean monthly temperature and mean daily minimum; Fh is the "helio factor" based on day length, latitude and global radiation, and Ht is the "heliothermic factor". Ft was held constant throughout the analysis. The available soil moisture is represented by the dryness factor ("facteur sécheresse") and is combined with Ht as shown in equation 2:

$$Ht \times Fs = CA \qquad (2)$$

where Ht is the heliothermic factor from Eq. 1, Fs is the monthly soil water balance and CA is the index of potential biomass productivity. CA is determined for each month and summed to obtain the annual value. Williams (1985) provides further detail on how the indices are calculated. We calculated CA under the 1961-90 normal climate and the 2020, 2050 and 2080 CGCM1-GG climate change scenarios. The index for the same levels of soil water-holding capacity were calculated for those above: 50, 100, 150, 200, 250 mm. The value of *CA* under the future climate scenarios was expressed as the proportion of the index under the 1961-90 climate normals.

Forest fire intensity

Potential head-fire intensity under current and future climate change scenarios was provided by a project recently completed at the Northern Forestry Centre, Canadian Forest Service, Edmonton, AB funded by the Prairie Adaptation Research Collaborative (PARC) (Kafka et al. 2001). Forest cover types from the forest inventory data were translated into fuel types used in the Canadian Forest Fire Behavior Prediction System (FBP), using a translation routine developed by fire management personnel from the Forest Protection and Fire Management Branch, Saskatchewan Environment and Resource Management, Prince Albert, SK (Frank, personal communication). The FBP predicts fire behaviour characteristics for each fuel type such as rate-of-spread, crown fire potential, fuel consumption and fire intensity. Inputs to the FBP are various components of the Canadian Fire Weather Index (FWI, Van Wagner 1987) which are determined from standard climate data. Kafka et al. (2001) used future climate inputs to the FWI system generated by the Canadian Regional Climate Model (CRCM, Caya and Laprise 1999) and then used the FBP system to determine head-fire intensities for the fuel types in each study area (Table 2) for both spring (March, April, May) and summer (June, July, August) fire seasons. The CRCM estimates future climate under somewhat different scenarios than does the CGCM1. The CRCM uses scenarios based on atmospheric CO_2 concentrations rather than specific time periods. Therefore, estimates of head fire intensity were provided for the $1xCO_2$ (i.e. current), $2xCO_2$ and $3xCO_2$ climate scenarios. The $2xCO_2$ scenario corresponds approximately to the 2050 CGCM1 time slice, and the $3xCO_2$ scenario corresponds approximately to the 2080 CGCM1 time slice. Results were expressed as the ratio of $2xCO_2$ to $1xCO_2$ and $3xCO_2$ to $1xCO_2$.

Table 2 FBP fuel types occurring in the three study areas and their key characteristics (adapted from Kafka et al. 2001).

Name	Designation	Characteristics
Spruce-lichen Woodland	C-1	Open stands with trees in dense clumps Black spruce branches extending to the forest floor Continuous reindeer lichen on the forest floor
Boreal Spruce	C-2	Moderately well-stocked Spruce crowns extending to near the ground Deep organic layer
Mature Jack Pine	C-3	Fully stocked stands of mature trees Live crown well above surface fuels Herb and shrub cover is sparse
Immature Jack Pine	C-4	Pure dense stands of immature trees Continuous vertical and horizontal fuels Heavy accumulations of dead and down woody fuels

Name	Designation	Characteristics
Boreal Mixedwood	—1/M-2	Moderately well-stocked stands of boreal coniferous and deciduous species Conifer crowns may extend to the ground Moderate shrub and herb cover Proportion of deciduous vs. coniferous species determines fire behaviour M1 - deciduous species are leafless (spring) M2 - deciduous species are green (summer)
Leafless Aspen	D-1	Pure semi-mature moderately well-stocked stands Ladder fuels absent Well developed shrub layer Continuous leaf litter
Jack Pine Slash	S-1	Continuous slash from mature jack pine stands Slash is usually 1-2 years old retaining up to 50% of foliage
Grass	O1	Continuous grass; fuel load is variable; degree of drying can influence fire behaviour

Insects

Assessment of susceptibility to insect infestation was limited to spruce budworm (SBW), jack pine budworm (JPBW) and forest tent caterpillar (FTC), as these are the major economically important forest pests in the western Canadian boreal forest (Fleming 2000, Volney and Fleming 2000). Susceptibility was determined subjectively based on expert opinion (Dr. Rory Mcintosh, provincial forest entomologist, Prince Albert, SK) and published data. We used forest inventory data to calculate the area of each map sheet that fell into a susceptible cover type based on species and age-class distribution. Because the age of trees partly determines their susceptibility to insect attack, we determined the area of each landscape in the Mature and Over-Mature (M&OM) class, i.e. older than the rotation age of 70 years for commercial forest species in Saskatchewan. We then combined that information with suggested shifts in forest species composition due to natural succession or following changes due to moisture stress or fire.

RESULTS

Moisture Stress

AET/PET for the study sites is shown in Figures 3-5. The effect of soil AWC is evident, particularly for the BF (Figure 3) and HB (Figure 4) landscapes. For these areas, AET is 80 - 90% of PET under the current climate, but decreases by 2080 to 60% for 50 mm AWC soils and 80% for 250 mm AWC soils. The ML landscape (Figure 5) is less affected, with AET/PET of 95-100% under the current climate, decreasing to 75-90% by 2080.

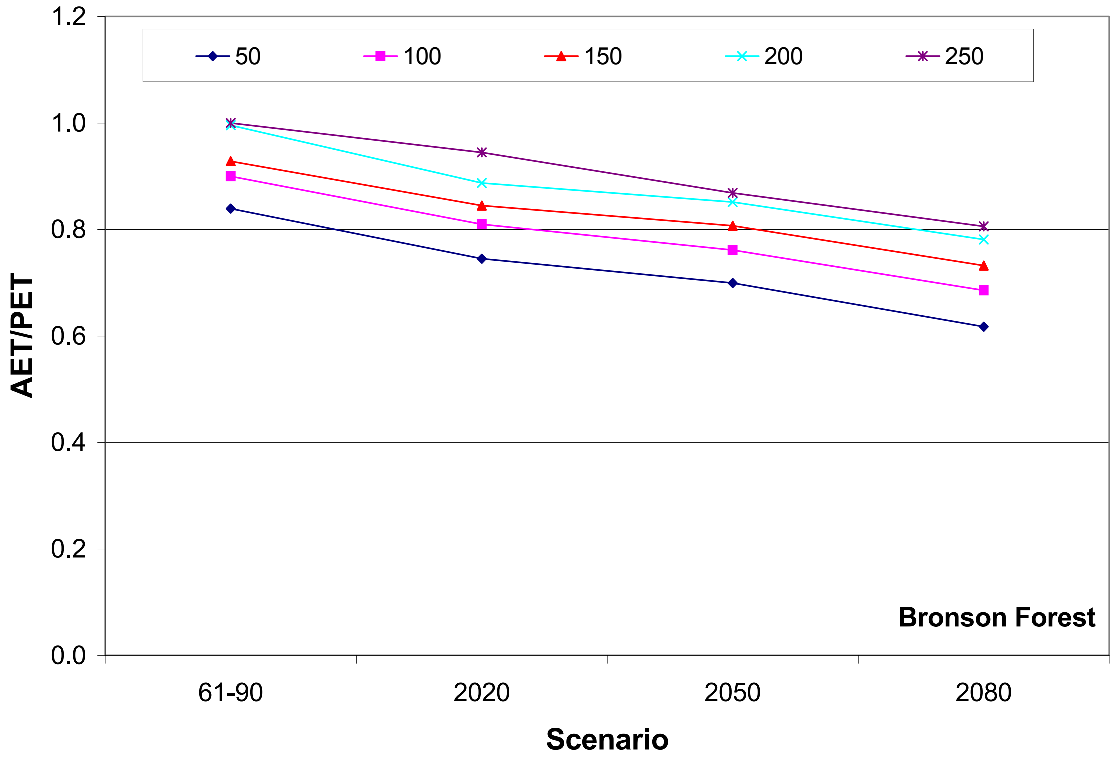

Figure 3 **Ratio of annual actual evapotranspiration to annual potential evapotranspiration for the Bronson Forest study area.**

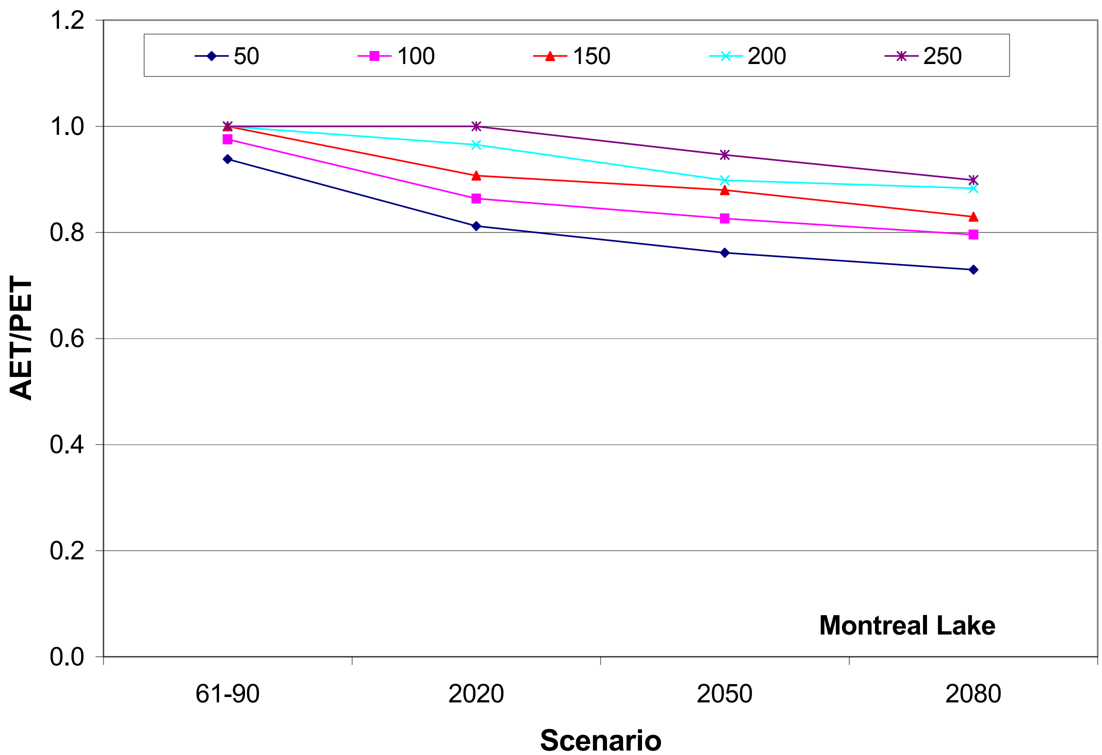

Figure 4 **Ratio of annual actual evapotranspiration to annual potential evapotranspiration for the Montreal Lake study area.**

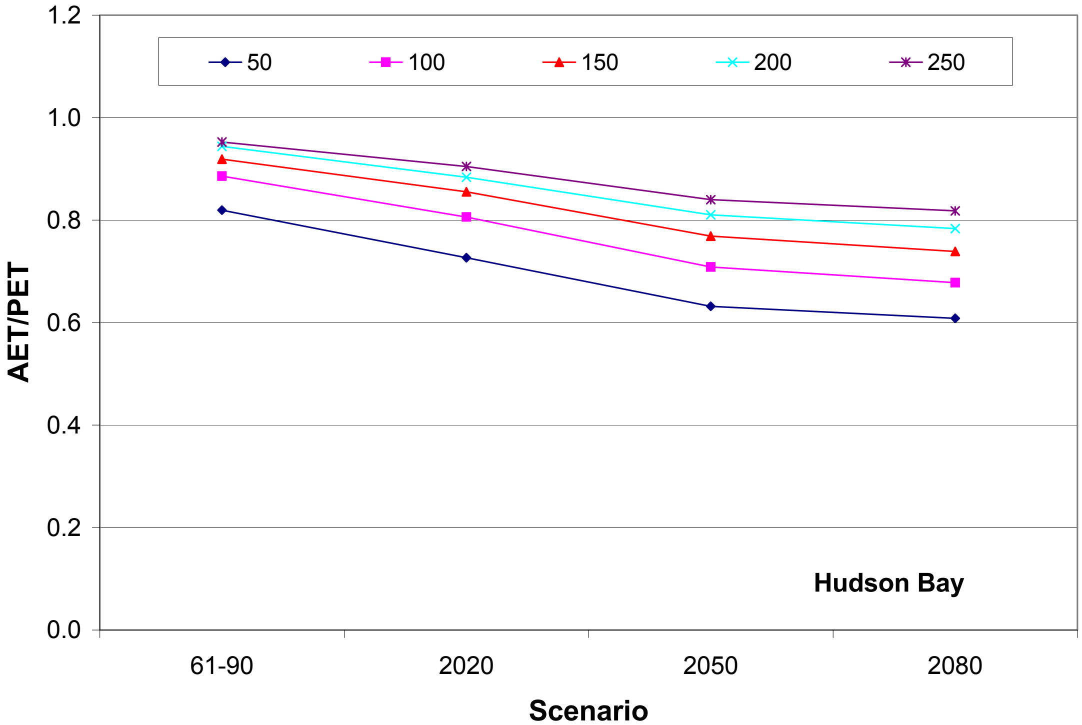

Figure 5 **Ratio of annual actual evapotranspiration to annual potential evapotranspiration for the Hudson Bay study area.**

Soil moisture deficits provide a better indication of the effects of moisture stress on forest vegetation. While the absolute deficit gives some indication of moisture stress, a more valuable index is the deficit as a proportion of total soil water holding capacity. The 1961-1990 normals indicated that the three study areas experience the greatest soil moisture deficits in August. Therefore, the August soil moisture deficit was used to examine the extremes under the future climates and examined how soil water holding capacity influenced soil moisture deficits. Soil moisture deficit as a proportion of AWC is shown in Figures 6-8. On the BF landscape (Figure 6), the soil moisture deficit is 100% of AWC by 2020 for soils with 50 mm AWC; i.e. soil moisture is exhausted. Deficits on soils with greater AWC reach between 60 and 80% of AWC in 2020; this increases to 70-95% of AWC by 2080. Deficits on soils with the highest levels of AWC (250 mm) reach 69% of AWC in 2080, resulting in only 31% of the potential soil water being available for tree growth. For the ML landscape (Figure 7), deficits are generally not as severe. On soils with 50 mm AWC, deficits reach 96% of AWC by 2020 and 100% by 2080. However, soils with higher levels of AWC do not reach as high a level of deficit as compared to BF. For soils with 100 mm AWC, the maximum deficit is 80% by 2080, while that on soils with 250 mm AWC only reaches 60% by 2080. On the HB landscape (Figure 8), moisture deficits reach high levels earlier as compared to the other sites. On soils with 50 mm AWC, soil moisture deficit is virtually always equal to soil AWC. For soils with higher AWC, by 2080 deficits are 79 to 100% of AWC, the highest of the three landscapes.

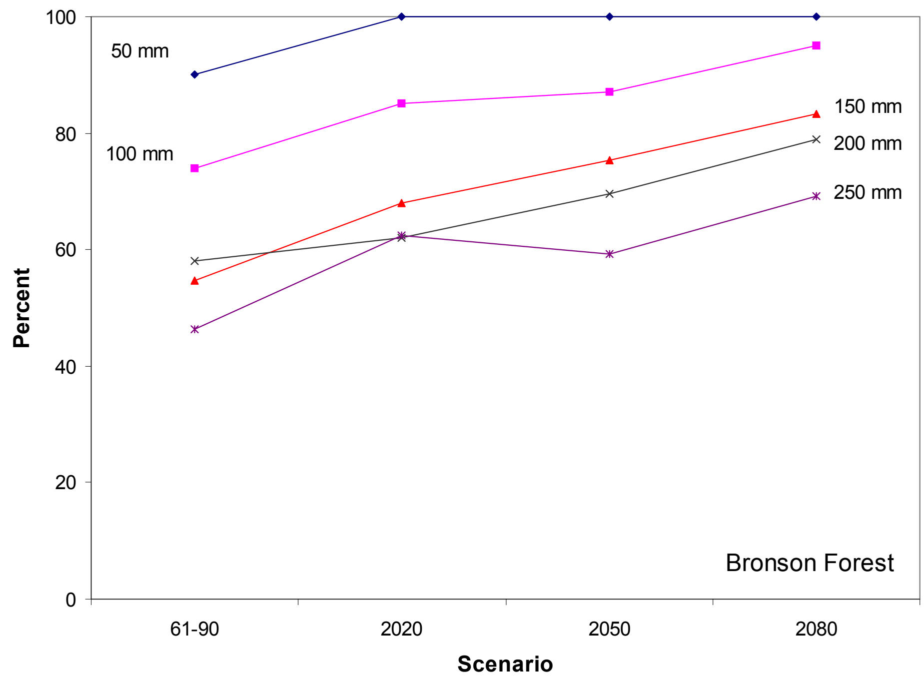

Figure 6 August soil moisture deficit as a percent of soil available water holding capacity for the Bronson Forest study area.

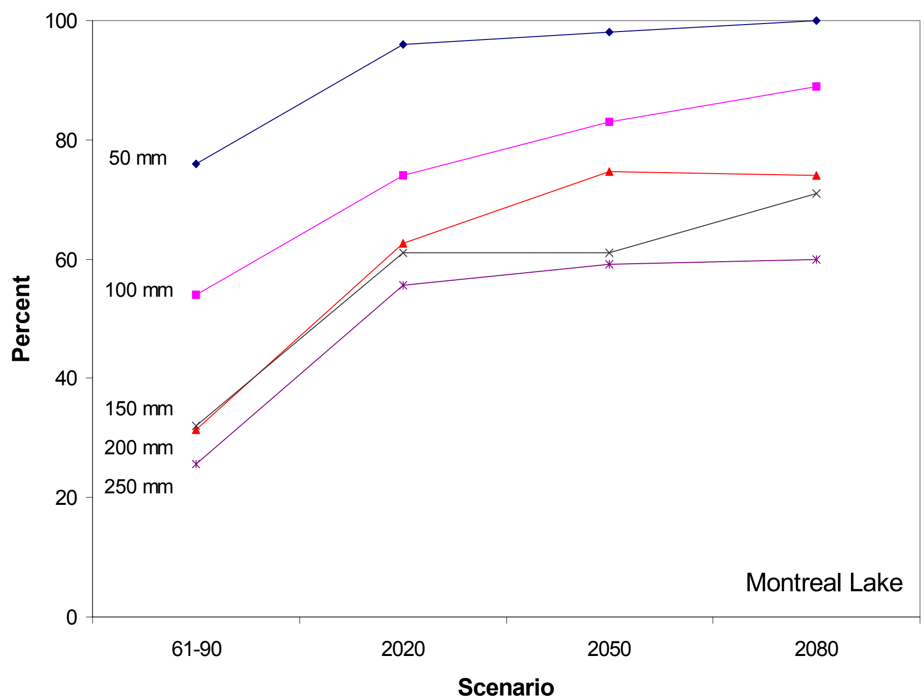

Figure 7 August soil moisture deficit as a percent of soil available water holding capacity for the Montreal Lake study area.

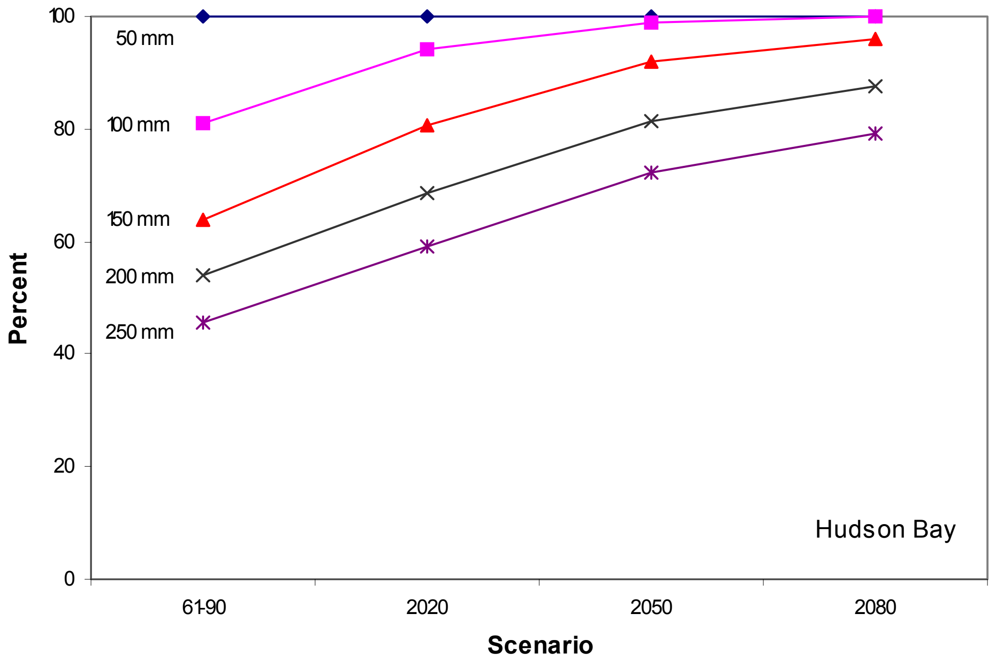

Figure 8 **August soil moisture deficit as a percent of soil available water holding capacity for the Hudson Bay study area.**

It appears that generally deficits on soils with > 150 mm AWC tend not to reach 100%, so that this level of AWC can be considered a threshold below which severe moisture deficits should be expected. With this in mind, we determined the extent of each landscape occupied by soils with less than 150 mm AWC (Table 3).

Table 3 **Area (ha) occupied by soils with < 150 mm AWC.**

	Area	
	(ha)	(%)
BF	6560	96.9
ML	5493	88.0
HB	1372	20

The BF landscape is made up almost entirely of soils with AWC of less than 150 mm and is also under high moisture demand as indicated by the value of AET/PET (Figure 3). While the ML landscape also has a large proportion of low water-holding soils, the overall moisture demand, as indicated by AET/PET (Figure 4), is less and therefore the incidence of drought stress should be lower. For the HF landscape, moisture demand is higher according to the value of AET/PET (Figure 5). While only 20% of the area is occupied by soils with low AWC, moisture deficits on soils with higher AWC is nearly the same as that on the drier soils in the BF landscape. In summary, of the three study areas, the BF and HB landscapes is most at risk of severe moisture stress under a future climate; the BF landscape's vulnerability is due to its soil characteristics, while that of the HB landscape is due to increasing climatic stress.

Productivity

The productivity index combines the effect of both temperature and moisture availability and indicates the effects of these factors on biomass production (Turc and Lecerf 1972). Figures 9-11 shows the values of this index for the three future scenarios as a proportion of that calculated under the 1961-1990 normals, for the five levels of AWC. For the BF landscape (Figure 9), productivity increased from present values for all but the 100 mm AWC soils, which decreased by about 10%. By 2050, only the soils with more than 150 mm AWC had increased levels of productivity. Productivity on soils with lower levels of AWC declined by up to 10%, and this declined increased to 20-25% by 2080. However, productivity on soils with more than 150 mm AWC remained at high levels, increasing by up to 35%. As shown in Table 3, the majority of the BF study area is made up of soils with less than 150 mm AWC, so the potential extent of productivity decline in this landscape is large.

For the ML landscape (Figure 10), by 2020 productivity remained the same on soils with 50 mm AWC, and increased between 10 and 30% on soils with higher levels of AWC. For soils with less than 150 mm AWC, productivity then declined through the 2050 and 2080 scenarios, with the driest soils experiencing a decline in productivity of 23%. Soils with 150 mm AWC experienced a decline in productivity relative to the 2020 scenario, but were still of higher productivity as compared to the 1961-1990 normals. Soils with higher AWC increased in productivity (250 mm) or remained the same (200 mm). By 2080, productivity on soils with 250 mm AWC had increased by nearly 40% over that of the 1961-1990 normals. About 88% of the ML landscape is occupied by soils with less than 150 mm AWC, so there is potential for declines in productivity on these drier sites similar in magnitude to that on the BF landscape.

For the HB landscape (Figure 11), the potential for decreases in productivity was the most pronounced, with uniform declines on all soils. Soils with less than 200 mm AWC experienced declines under all scenarios; e.g. productivity on soils with less than 150 mm AWC was reduced by 44% by 2080. Soils with greater than 150 mm AWC experienced an increase in productivity under the 2020 scenario, but this increase began to disappear by 2050 and had been eliminated by 2080. By this time, productivity on the wetter soils had returned to the 1961-1990 levels (250 mm AWC) or declined by about 15% (200 mm AWC). Although potential productivity declines are large on low AWC soils, these comprise only about 20% of the HB landscape. Due to the larger extent of high AWC soils, productivity under the future climate in this study area will likely increase somewhat and then return to within 15% of current levels.

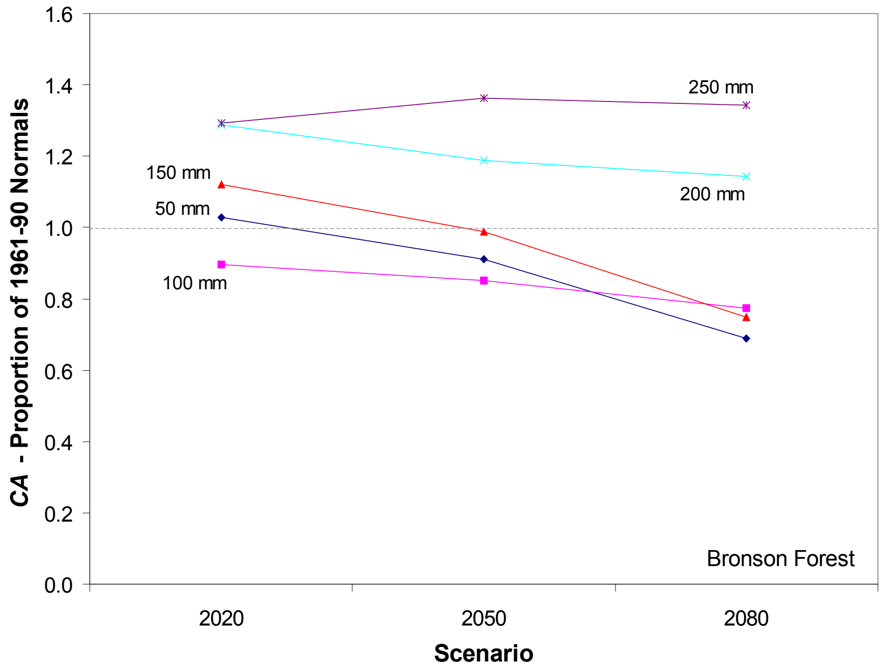

Figure 9 **Future biomass productivity index (CA) as a proportion of that under the 1961-1990 normal climate for the Bronson Forest study area.**

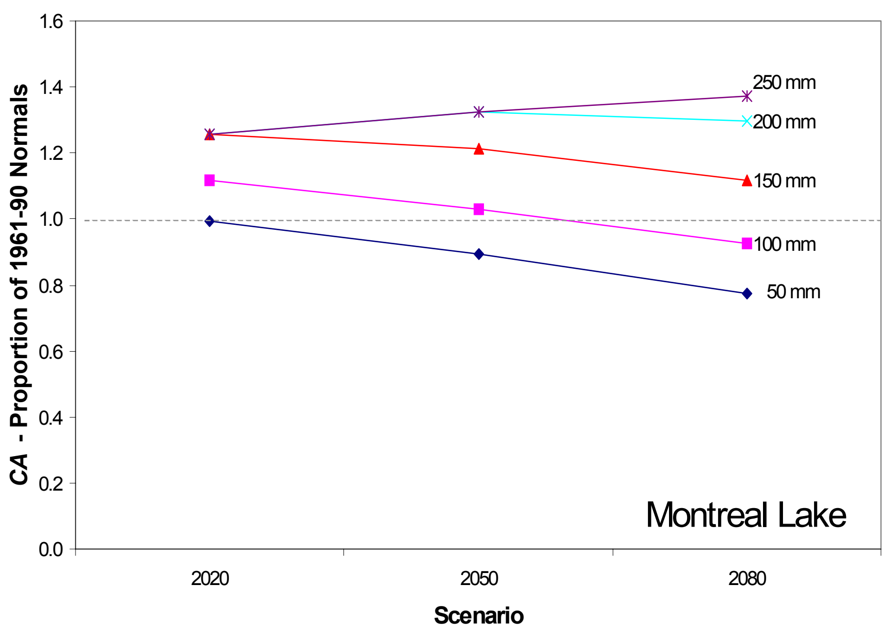

Figure 10 **Future biomass productivity index (CA) as a proportion of that under the 1961-1990 normal climate for the Montreal Lake study area.**

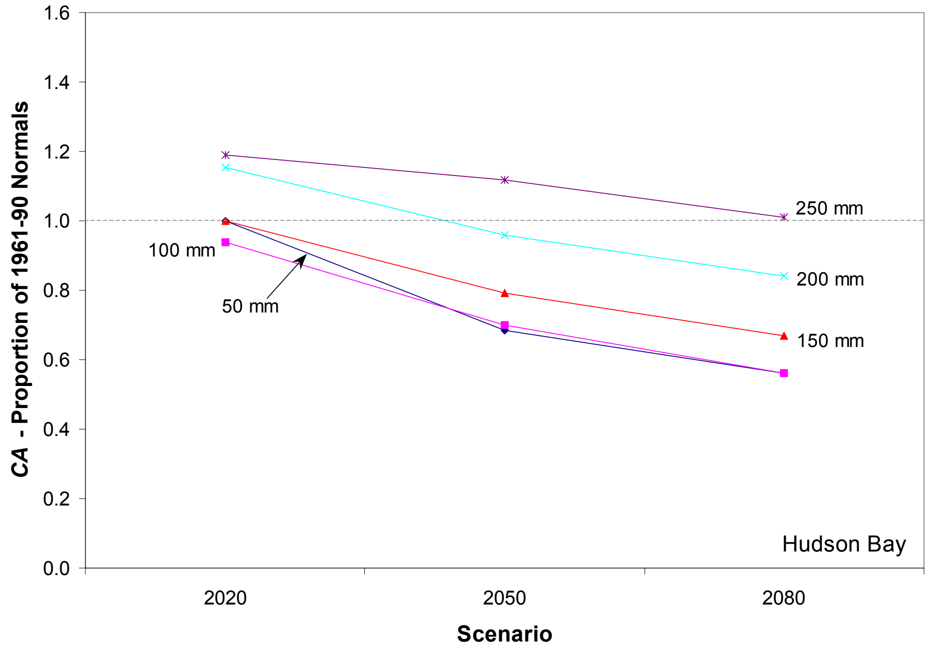

Figure 11 Future biomass productivity index (CA) as a proportion of that under the 1961-1990 normal climate for the Hudson Bay study area.

The Heliothermic component (*Ht*) of the productivity index is shown in Figure 12. Since the radiation component of *Ht* was held constant, Figure 12 represents the effect of change in temperature among the three scenarios. There is a trend of increasing temperature for all study areas, with the increase in the BF landscape the greatest, that for HB intermediate that for ML the least. It appears that productivity will increase with increasing temperature given sufficient soil moisture availability; as shown above; however, soil moisture deficits are potentially severe on these landscapes, particularly in the BF study area.

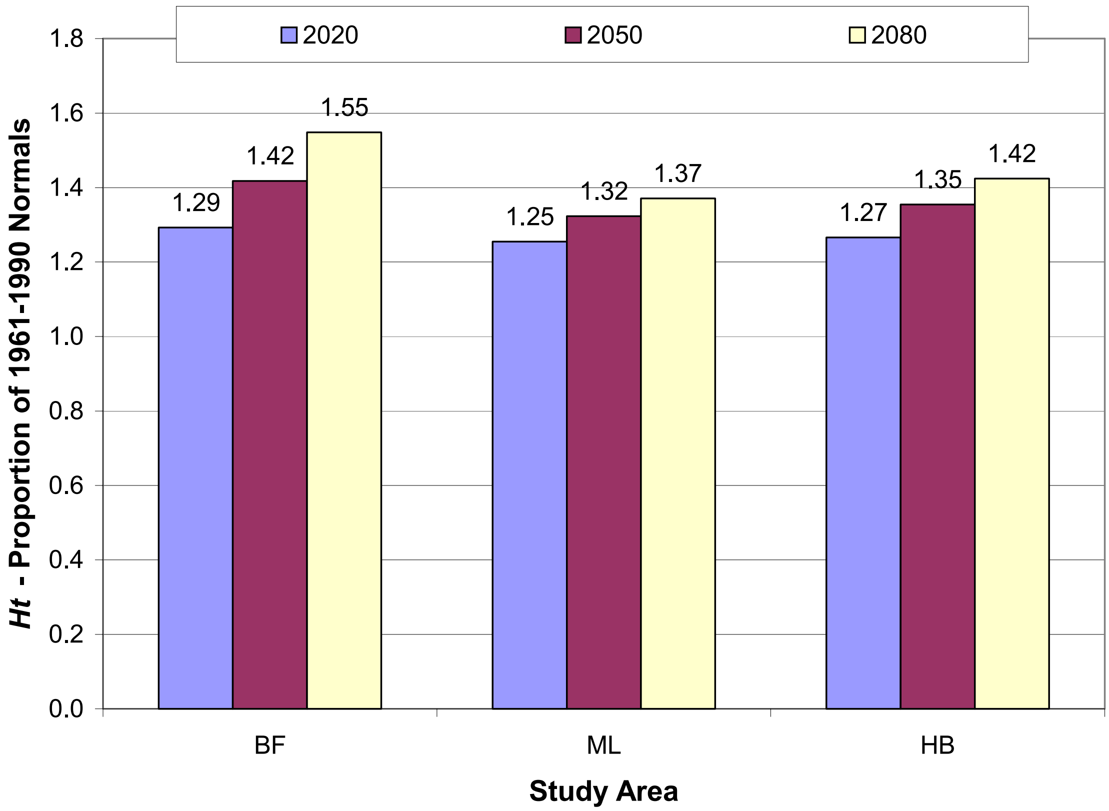

Figure 12 Heliothermic component (Ht) of the biomass productivity index for the three study areas. BF, Bronson Forest; ML, Montreal Lake; HB, Hudson Bay.

Forest Fire Intensity

Head fire intensity under current and future climate varied among the three landscapes for both spring and summer fires seasons. Figures 13-18 show the changes in fire intensity for the 2xCO_2 and 3xCO_2 scenarios during the spring and summer fire seasons. For the BF landscape, spring head fire intensities increase for all fuel types except C1 and O1 under the 2xCO_2 (Figure 13). The increase is 30-45% for the C2-C4 fuel types, nearly 25% for the M1/M2 fuel type, and 15% for the D1 and S1 fuel types. Fire intensity values declined by 12% and 23% for the O1 and C1 fuel type, respectively. Under the 3xCO_2 scenario, potential intensities increased 80% for the C1 fuel type, and 35% for the O1 fuel type. Increases for the other fuel types were less, ranging between 3% for the C4 type and 27% for the C3 type. For the summer fire season on the BF landscape, fire intensities remained nearly the same or decreased slightly. The exceptions were the C1 and O1 fuel types; for the C1 fuel type, fire intensities increased 12 and 35% for the 2xCO_2 and 3xCO_2 scenarios, respectively, while for the O1 fuel type, intensities increased 9% under the 2xCO_2 scenario and 21% under the 3xCO_2 scenario.

The results for the ML landscape were quite different. Potential fires intensity levels declined under both scenarios for all fuel types in the spring fire season (Figure 15). For the 2xCO_2 scenario, declines were largely between 20% and 25% for most fuel types, but were as large as 53% for the

C3 fuel type and 44% for the C1 fuel type. Under the $3xCO_2$ scenario, declines were less, falling in the 5-10% range for most fuel types. For the summer fire season on the ML landscape, there were large increases in potential fire intensity (Figure 16). Values increased by 51% in the C1 fuel type and 42% for the O1 fuel type. Increases in the other fuel types ranged from 16% for the C3 fuel type to 7% for the C2 fuel type. Fire intensity declined under the $3xCO_2$ scenario for all fuel types. Declines for most fuel types were in the 20-25% range, with larger declines of 42% for the C1 fuel type and 38% for the C3 fuel type.

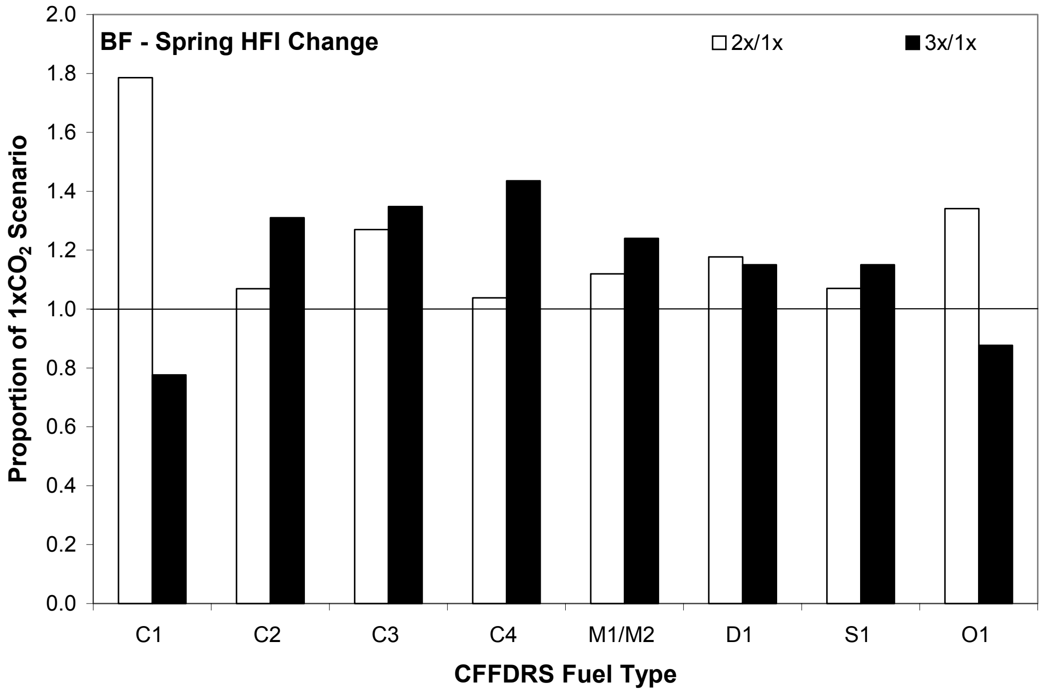

Figure 13 **Future spring fire severity as a proportion of that under the 1961-1990 normal climate for the Bronson Forest study area.** See Table 2 for definitions of Canadian Forest Fire Danger Rating System (CFFDRS) fuel types.

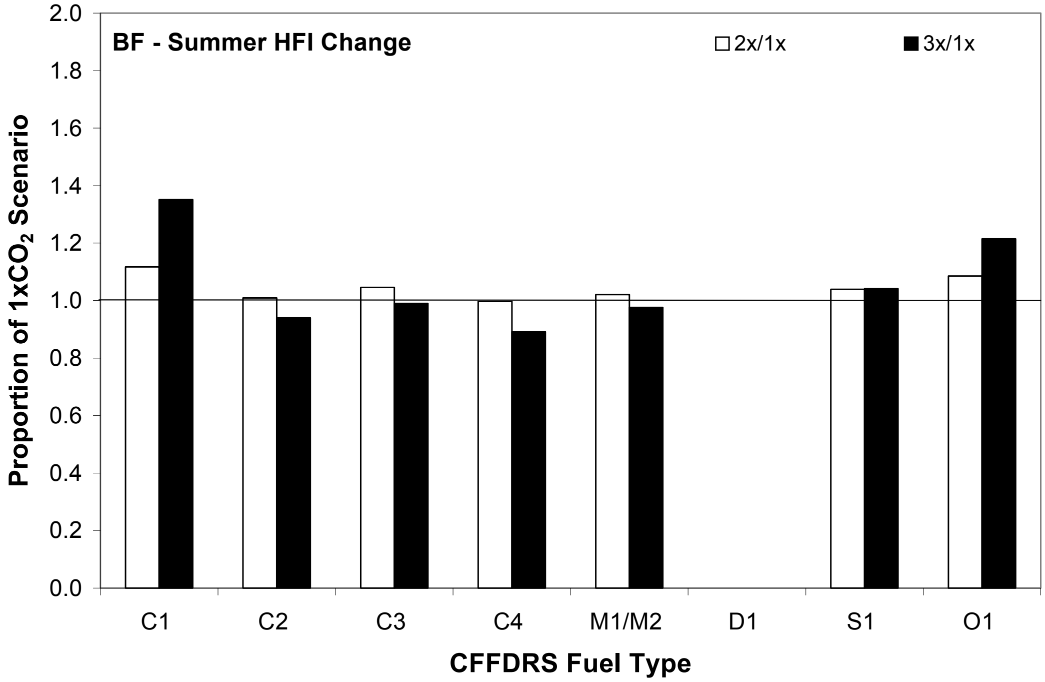

Figure 14 Future summer fire severity as a proportion of that under the 1961-1990 normal climate for the Bronson Forest study area. See Table 2 for definitions of Canadian Forest Fire Danger Rating System (CFFDRS) fuel types.

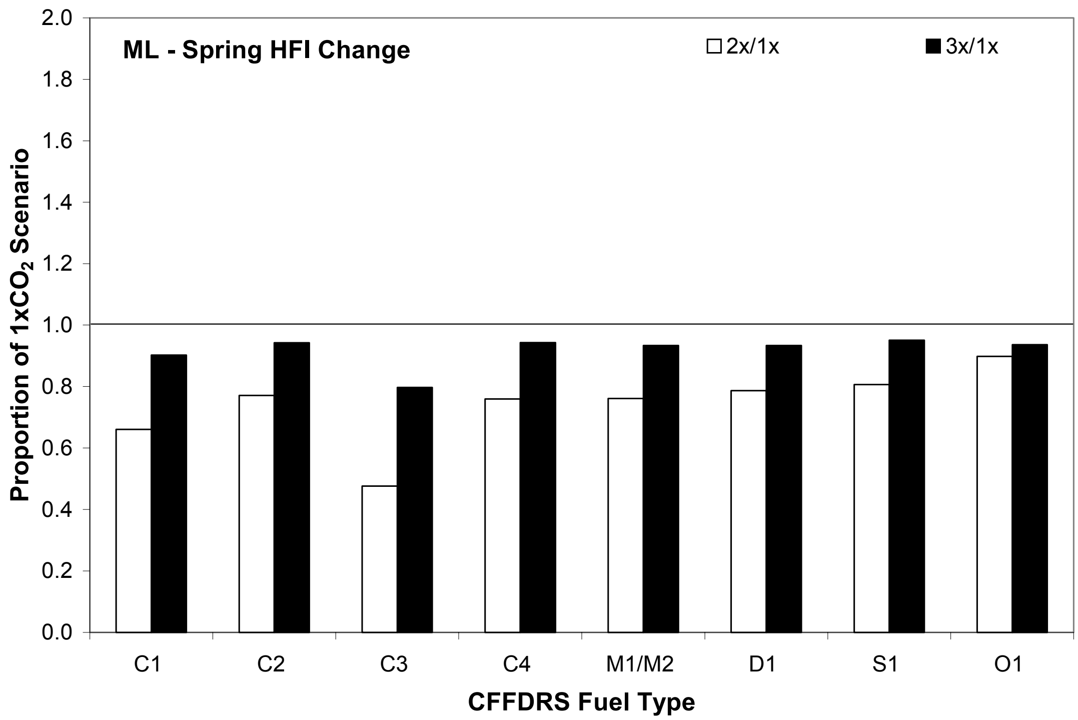

Figure 15 Future spring fire severity as a proportion of that under the 1961-1990 normal climate for the Montreal Lake study area. See Table 2 for definitions of Canadian Forest Fire Danger Rating System (CFFDRS) fuel types.

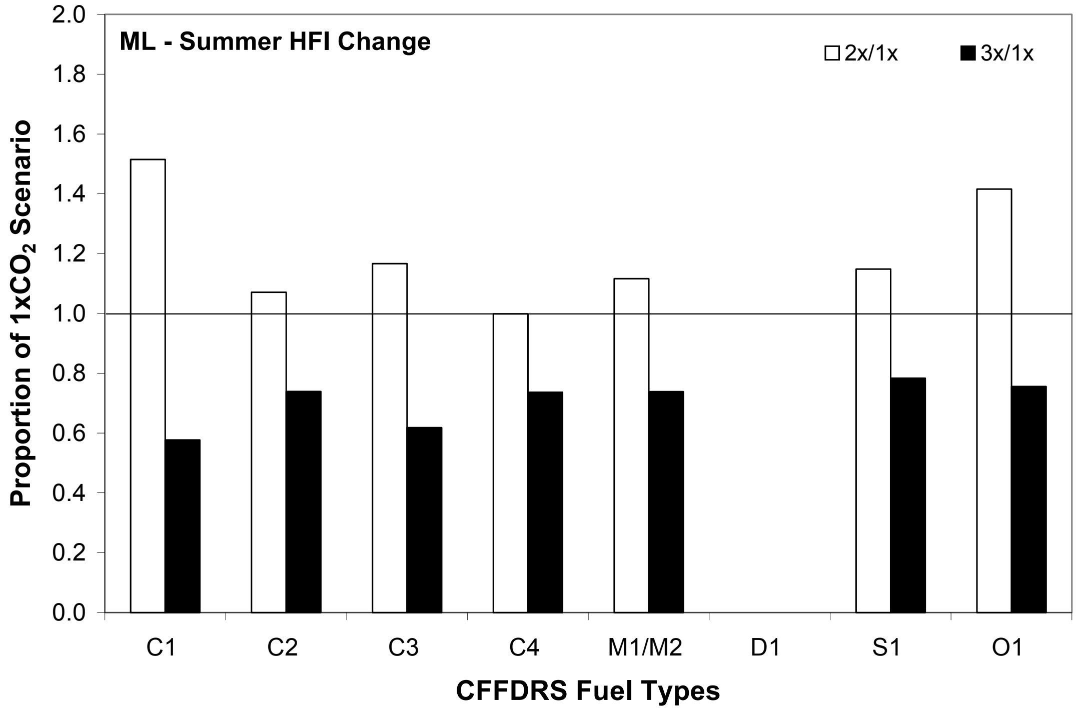

Figure 16 **Future summer fire severity as a proportion of that under the 1961-1990 normal climate for the Montreal Lake study area.** See Table 2 for definitions of Canadian Forest Fire Danger Rating System (CFFDRS) fuel types.

For the HB landscape, potential spring fire intensities decline for all fuel types under the $2xCO_2$ scenario, ranging from nearly 60% in the C1 fuel type to 12% for the C4 fuel type. Under the $3xCO_2$ scenario, intensities increase from 4 to 36%, with increases of more than 20% for the C2, C3 and C4 fuel types. For the summer fire season, fire intensity declines under the $2xCO_2$ scenario for all fuel types. Declines were between 10 and 20% for the C2-C4, M1/M2 and S1 fuel types, increasing to 30-40% for the C1 and O1 fuel types.

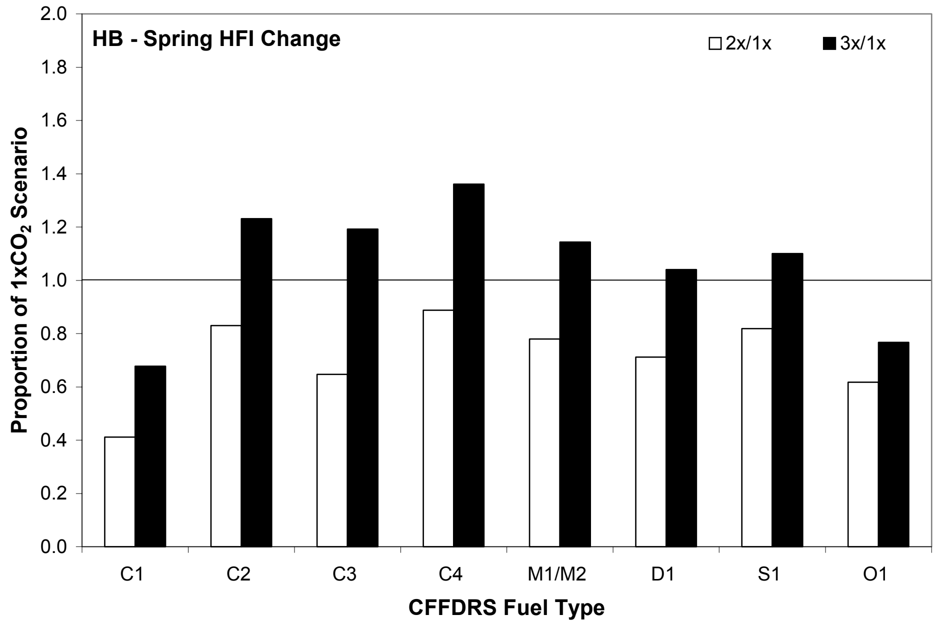

Figure 17 **Future spring fire severity as a proportion of that under the 1961-1990 normal climate for the Hudson Bay study area.** See Table 2 for definitions of Canadian Forest Fire Danger Rating System (CFFDRS) fuel types.

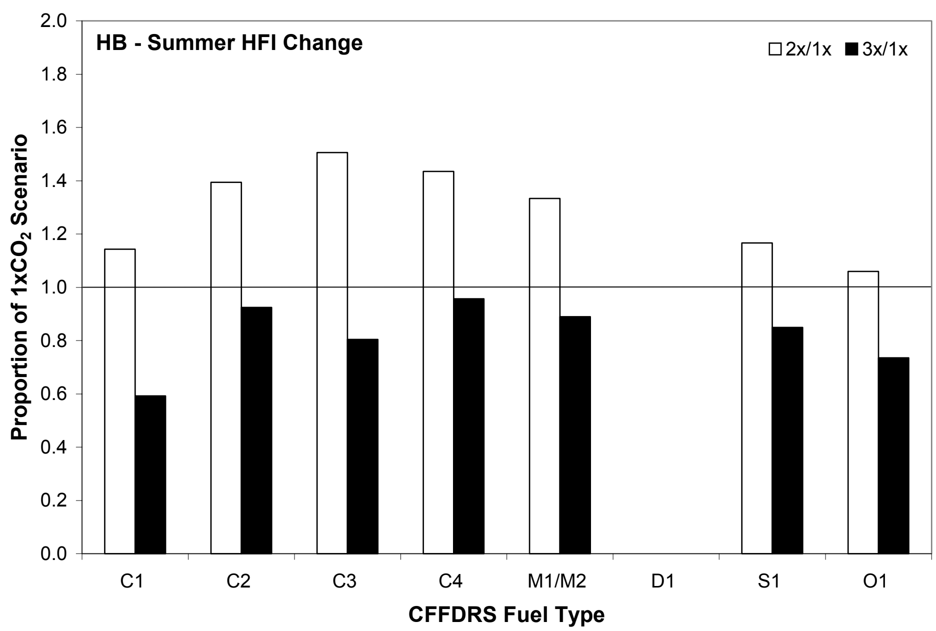

Figure 18 **Future summer fire severity as a proportion of that under the 1961-1990 normal climate for the Hudson Bay study area.** See Table 2 for definitions of Canadian Forest Fire Danger Rating System (CFFDRS) fuel types.

It is important to recognize the areal extent of the various fuel types and how they vary among the three landscapes (Table 4). While some increases in future fire intensity are large, in some cases these increases apply to a very small proportion of the landscape and would be relatively unimportant to future fire management. For the BF landscape, nearly all the forest area (94%) falls into the D1 (Deciduous) fuel type. For this fuel type, only the spring fire season is important, as the summer condition is such that flammability is extremely low due to the large amount of green foliage with high moisture content (Forestry Canada Fire Danger Group 1992). Figure 13 indicates that potential fire intensity increases by 18% and 15% under the 2xCO_2 and 3xCO_2 scenarios, respectively. Fuel type M1/M2 occupies 4.5% of the area, and will experience increases of 12% and 24% under the 2xCO_2 and 3xCO_2 scenarios, respectively. For the summer fire season, fuel type D1 does not occur, and fire intensity in the M1/M2 fuel type remains within 5% of the 1xCO_2 values (Figure 14). We conclude that spring fire danger in the deciduous forest types is an area of concern for this landscape.

For the ML landscape, coniferous (C2-C4) and mixedwood (M1/M2) fuel types dominate, with some area in D1. Spring fire intensity values decline among all fuel types under both scenarios (Figure 15), while those in summer increase for C2, C3 and M1/M2 fuel types (Figure 16). Based on the areas shown in Table 4, potential summer fire intensity in C3 and M1/M2 should be of concern to managers in this landscape.

In the HB landscape, M1/M2 and D1 comprise the majority of the area. For these fuel types, potential intensity declines in spring under the 2xCO_2 scenario but increases under the 3xCO_2 scenario, with intensity in the M1/M2 fuel type increasing by 14% and that for D1 increasing by 4%. In the summer fire season, potential fire intensity increases by 33% in the M1/M2 fuel type under the 2xCO_2 scenario and all levels of fire intensity decline under the 3xCO_2 scenario. The cause for concern in this landscape would be summer fire intensity for the M1/M2 fuel type under the 2xCO_2 scenario, although relative to the other study areas, the future fire severity on the HB landscape will not change significantly.

Table 4 Area (%) of each fuel type for the three study areas.

Fuel type	BF	ML	HB
C1	0.2	0	0
C2	0.4	19.1	5.4
C3	0.8	10.2	0
C4	0.1	18.1	0
M1/M2	4.5	38.2	34
D1	93.9	14.4	53.2
S1	0.0	0.0	7.4
O1	0	0	0

Insects

The risk of insect attack related to climate change impacts is difficult to predict. There is a lack of quantitative information on the relationship between climate factors (e.g., temperature, precipitation) and insect populations, and the ecological impacts of climate change on insects is even more difficult to predict. We looked at tree species composition and age class structure and made a subjective assessment of risk associated with climate change among the three study areas. Figure 19 shows the age-class distribution for the three study areas and Figure 20 shows the area of stands designated Mature and Over-Mature (M&OM) in the provincial inventory. This refers to stands that are beyond the rotation age, i.e. 80 years and older. The BF landscape is composed primarily of stands in the 60 year age class of which 97% is aspen. These stands date from large fires in the 1940s and 1950s. Forest tent caterpillar (FTC) has been a common occurrence in this area in the past and, in association with drought, has caused large areas of aspen decline in this part of Saskatchewan (Hogg and Schwarz 1999).

The FTC has been shown to attack stands in nearly all age classes, so even though these stands are fairly young (Figure 20), they are still highly susceptible. Under future climate scenarios, trees will likely experience increased drought stress and productivity decline as shown above, making them more susceptible to insect attack (Mattson and Haack 1987). In addition, increased fire disturbance will tend to drive forest composition toward early successional species, which in this area is most likely to be either aspen or jack pine. Jack pine stands currently occupy only about 2% of the landscape area, so aspen is the most likely species to return after fire. Therefore it is likely that the majority of this landscape will be dominated by aspen in the future and will be at high risk of attack by FTC.

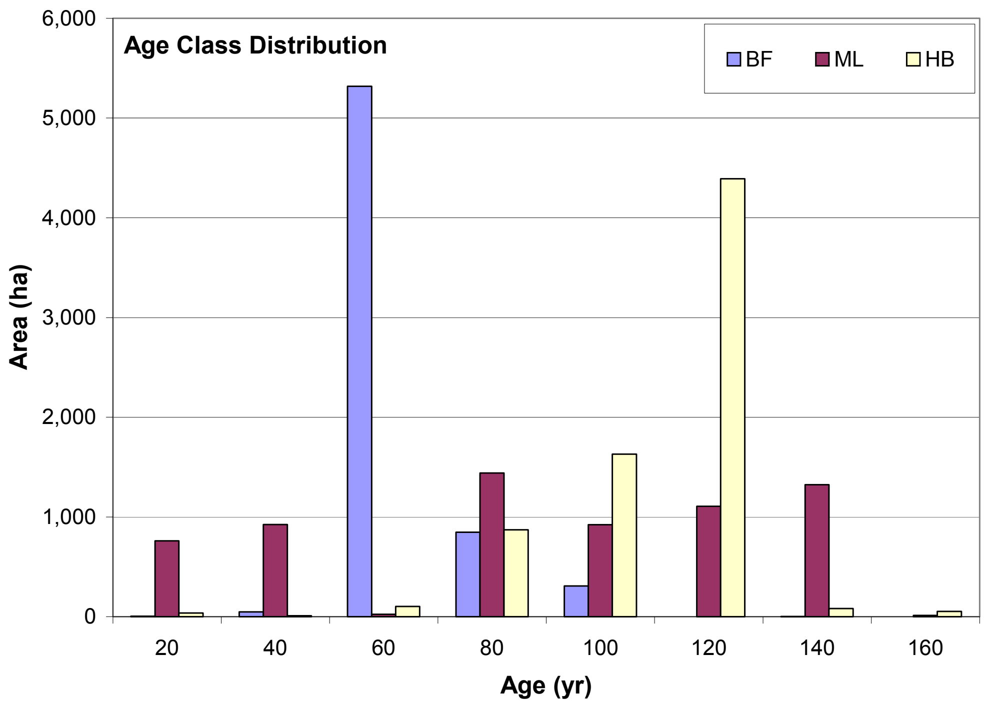

Figure 19 Age class distribution for commercial species in the three study areas from the Saskatchewan forest inventory. Study area designations as in Figure 12.

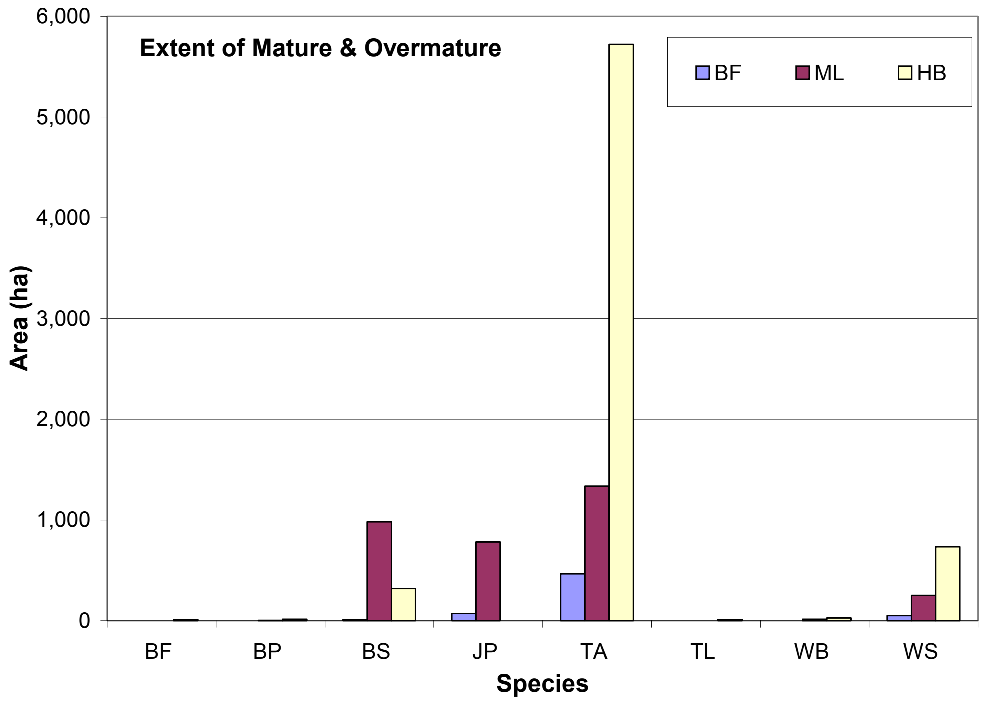

Figure 20 **Areal extent of Mature and Over-Mature age classes for commercial species in the three study areas from the Saskatchewan forest inventory**. Species designations: BF, balsam fir; BP, balsam poplar; BS, black spruce; JP, jack pine; TA, trembling aspen; TL, tamarack larch; WB, white birch; WS, white spruce. Study area designations as in Figure 12.

The situation for the ML landscape is quite different. Figure 19 indicates that the age-class distribution is relatively well balanced. Figure 20 shows that there are stands of M&OM black spruce, jack pine and aspen, each comprising 10-13% of the landscape, and some stands of white spruce, about 2.5% of the landscape. Areas of jack pine occurring on soils with low AWC may be at risk of attack by jack pine budworm (Volney and Fleming 2000). As shown above, about 88% of this landscape contains drought-prone soils (Table 3), so this could be an issue of major concern. These areas typically return to jack pine following fire (Kabzems et al. 1986), so the future species composition will probably cause this level of susceptibility to be maintained. The area of M&OM aspen is vulnerable to FTC, representing about 13% of this landscape. White spruce and to a lesser extent black spruce are prone to SBW attack (Blais 1985) but occupy relatively small areas of this landscape. The vulnerability of jack pine on drought-prone soils to JPBW is the most important concern on this landscape, with FTC attack in aspen of additional concern.

The HB landscape is dominated by M&OM stands in the 80-120 year age-classes (Figures 19 and 20). These stands are primarily aspen with some black spruce and white spruce. As with the BF landscape, the HB area is susceptible to FTC outbreaks due to the large area of mature aspen, although the impacts of drought may be less due to the smaller area of drought-prone soils, about 20% of the landscape (Table 3). Older stands of white spruce are vulnerable to SBW but occupy a small area within this landscape. In this landscape soils with greater than 200 mm AWC represent the sites of greatest productivity both now and in the future (see section above on potential biomass productivity). These more productive sites often support mixedwood stands comprising various mixtures of aspen and white spruce (Kabzems et al. 1986). Succession usually progresses from a stand with an aspen overstory and a white spruce understory to one with a spruce overstory as the aspen die and release the spruce. On these sites, the current aspen-dominated stands will likely succeed to white spruce, creating a potential host for SBW. However, the relatively high AWC and greater productivity may allow the trees to avoid drought stress and resist the impacts of SBW.

In summary, the BF landscape has the highest level of vulnerability to climate change impacts when climatic, soils and vegetation influences are jointly considered. This landscape is subject to high climatic moisture demand, is underlain by low water holding capacity soils, and has large areas of tree species vulnerable to insect attack and subject to increasing fire severity. The HB landscape is climatically vulnerable but these impacts will be somewhat buffered by the large areas of high soil water holding capacity and more productive forest communities. The ML landscape is less vulnerable climatically but has some degree of risk associated with low water holding capacity soils.

DISCUSSION

Our results show clearly that the impacts of changes in climate will be strongly modified by local landscape conditions. In our study areas soil water holding capacity is extremely important in modifying the broader effects of climate on water supply and potential productivity. These modifications result in widely different ecosystem impacts under relatively similar climate change scenarios. For example, the HB landscape is the most severely affected according to the climate scenario data, while the vulnerability of the BF landscape is greater than that of the others when soil AWC is taken into account. We also found that landscapes in the forest-grassland ecotone will experience greater moisture deficits and productivity declines than those further from this boundary, although these impacts will be modified by local soil factors. Hogg (1994) found that the forest-grassland boundary in the prairie provinces was closely aligned with the value of a moisture index at which potential annual evapotranspiration exceeded annual precipitation. He also indicated that his climatic analysis did not include soil moisture availability, but suggested that this factor was also an important determinant of local soil moisture stress.

Local vegetation age structure and composition will also modify climatically-driven impacts of disturbance on forest ecosystems. Vulnerability of the study sites to fire and insect attack differed significantly due to both current vegetation and assumptions about how forests will change due to succession. Local vegetation and fuel characteristics play a large role in determining fire behaviour in addition to the effects of climate and weather (Forestry Canada Fire Danger Group 1992) and must be considered in analyses of climate change impacts on disturbance regimes. Weber and Flannigan (1997) suggest that the short-term impacts of changes to disturbance regimes have the potential to dwarf the long-term direct impacts of climate change on forests.

The interactions between various disturbance agents must be explicitly included in analyses of climate change impacts. Many studies have shown that drought predisposes forest stands to insect attack (e.g. Mattson and Haack 1987, Ayers and Lombardero 2000, Hanson and Weltzin 2000). Our study landscapes include large areas in which drought may be severe and which contain extensive stands of trees susceptible to insect pests (e.g. aspen on drought-prone soil in the BF landscape). These areas should be of concern and studied carefully for the early signs of climate change, which is perhaps already occurring (Hogg and Schwarz 1999). Similarly, insect attack can predispose conifer stands to increased levels of fire severity (Stocks 1987), so landscapes vulnerable to e.g. SBW may also be at increased risk of fire disturbance. Finally, many studies have established the link between drought and increased fire severity (Flannigan et al. 2000, 2001) and this needs to be a major factor in studies of future climate change impacts.

These results suggest that multiple spatial and temporal scales must be considered in the analyses of climate change impacts. Output of GCMs is produced on a grid approximately 400 x 400 km, a spatial scale much broader than that typical of variation in soils, vegetation and impacts of both human and natural disturbance. Downscaling techniques (Carter et al. 1999) can provide a grid of climate change at higher resolution (e.g. 10 x 10 km) but is still not consistent with scales of local ecosystem variation. Analyses of climate change effects must address variation at a range of scales and explicitly address the interaction among scales in order to predict likely ecosystem impacts. Similarly, temporal scales vary between climate processes and ecosystem function. Current GCM output provides data at 30-year time slices, whereas ecosystems experience the impacts of disturbance at scales of days (forest fires) to years (insect attack). These scale-dependent inconsistencies must be recognized and included in future climate change impact studies.

Our results underscore the importance of ecotones in the study of climate change impacts. Previous work indicates that ecotones may be sensitive indicators of climate change (Hogg 1994, Hogg and Hurdle 1995, Camill and Clark 2000) but that in some cases changes may be subtle and require relatively sophisticated analytical techniques. In particular, ecotones involving forested biomes present some distinct challenges. A major factor is the life-span of trees. Adult trees are robust in the face of environmental change and will not necessarily die immediately as climatic change occurs (Hanson and Weltzin 2000). A simple study of population change or a mortality survey will probably miss much of the early climate change impact, so that more sophisticated analytical techniques such as tree-ring analysis will probably be required (Noble 1993, Hogg and Schwarz1999). In addition, time lags and non-linear ecosystem responses have been found in analyses of past climatically-induced change in ecotones (Camill and Clark 2000), so that simple cause and effect relationships may be difficult to observe in the short term. However, this underscores the importance of establishing detailed, long-term monitoring networks now, in order to determine the baseline conditions against which future changes can be measured. An important example is the Climate Change Impacts on the Productivity and Health of Aspen (CIPHA) project which is establishing a series of long-term monitoring plots across central Canada. Its purpose is "to provide early detection of climate change impacts through monitoring of biomass, growth, and health of aspen forests in climatically sensitive areas in the western boreal forest and aspen parkland". (See the project web site http://nofc.cfs.nrcan.gc.ca/cipha/html/intro_e.html for further details).

Fragmentation is already a concern for forest land managers in the prairie provinces due to road construction, past unsustainable forest harvesting practices, agricultural clearing and urban expansion. Fragmentation has important impacts on natural forest regeneration, wildlife habitat and recreational opportunities (Harris 1984). An important implication of climate change for ecotone areas is an increase in fragmentation (Neilson 1993) . As the environment is increasingly stressed by e.g. drought, tree species are increasingly isolated in topographic positions on the landscape in which moisture is available (Hanson and Weltzin 2000, Clark et al. 2001). As we showed above, this is a potentially large problem in some of our study landscapes. As the drought stress becomes stronger there are fewer locations where trees have adequate moisture to grow and reproduce, resulting in increased fragmentation, decreased connectedness and increasing negative impacts on wildlife habitat, particularly for species with large home ranges (e.g. large ungulates) (Noss and Csuti 1997). In addition, habitat fragmentation may restrict or prevent the ability of species to migrate in order to track the changes in their environment (Pitelka et al. 1997).

Fragmentation also interacts with other disturbance processes. Roland and Taylor (1997) reported that populations of FTC increase with increased forest fragmentation in northern Alberta. The mechanism for the increase was determined to be an decline in parasitic fly populations in fragmented forest. The parasitoids caused a decline in FTC populations of up to 50% in intact forest as compared to populations levels in fragmented forest. Rothman and Roland (1998) reported that infection of FTC larvae by a viral pathogen was greater in contiguous forest, and that the probability of FTC larvae surviving infection dropped from *ca.* 90% in a fragmented forest to *ca.* 50% in intact forest. These studies agree in showing that levels of FTC are up to 50% greater in fragmented forest.

As shown above, climate change alone can exacerbate the vulnerability of forests to FTC, and fragmentation would be an additional factor in increasing that vulnerability. Forest fragmentation also affect the nature of fire disturbance regimes (Weber and Flannigan 1997). Forest fragmentation affects fuel arrangement and continuity, changing the patterns of forest fires. Fragmentation may slow the spread of crown fires which require contiguous conifer tree canopies (Johnson 1992). Alternatively, the open conditions of fragmented forests may allow greater grass biomass to develop, thereby providing increased fine fuel loading and fuel connectivity from one forest stand to the next; this has been shown in connection to historic land clearing and debris fires in the forest-agriculture fringe area in central Saskatchewan (Weir and Johnson 1998). In addition, post-fire forest regeneration could be limited by fragmentation if seed sources are distant from areas recently burned (Johnston 1996).

One of the goals of this study was to determine forest ecosystem impacts of climate change using easily available data and techniques. However, these are not efficient ways in which to integrate complex aspects of ecosystem function and structure (e.g. water balance, productivity, disturbance regimes). Modelling approaches exist that either currently or in the future will have the capacity to mechanistically model these ecosystem attributes. There have been several recent reviews of models appropriate for determining the impacts of climate change on forest (and other) ecosystems, and we outline these results here. While significant time is required to adequately parameterize these models, and input data (especially climate data which are often difficult to obtain), there seems to be little doubt that the future of climate change impact analysis lies in this approach. Neilson and Running (1996) separate climate change impact models into biogeography and biogeochemistry models.

Biogeography models capture the statistical relationships between current vegetation types and climate factors such as temperature, precipitation, growing degree days, etc. Biogeochemistry models predict vegetation productivity based on representations of cycles of water, carbon and nutrients. Output from GCMs is then used to drive the model which predicts the distribution of vegetation types under the future climate. Biogeography models are based on current or past relationships between vegetation and climate, but it is unlikely that these relationships will hold under future climates (Solomon and Kirilenko 1997). Biogeochemical models simulate ecosystem function but do not recognize the relationship between location and vegetation type and do not simulate vegetation population dynamics. The current direction in vegetation modelling for climate change impacts is to couple the two model types into a hybrid called a dynamic global vegetation model (DGVM). Peng (2000) provides a recent review of DGVM development, tracing the history of vegetation modelling from Koeppen's early climate-based classification to recent DGVMs that are coupled with GCMs resulting in an integrated geosphere-biosphere modelling approach. Current uses of DGVMs include simulating global vegetation distribution, productivity and biomass under transient climate change scenarios; coupling DGVMs with process-based biogeochemical models; and coupling DGVMs with GCMs for simulating vegetation feedbacks on climate (Peng 2000).

Recently, a number of model inter-comparisons have been carried out to explore strengths and weaknesses among various model approaches. Cramer et al. (2001) report on a comparison among six DGVMs which were used to investigate the interactions between ecosystem carbon and water exchange with vegetation dynamics. The DGVMs were coupled with the recent GCM developed by the Hadley Centre (HadCM2-SUL). The models all predicted an increase in net primary productivity (NPP) up to about 2030 primarily due to increased atmospheric concentration of CO_2; however, this fertilization effect declines after 2030 due to the saturation of the carbon response. This is an example of how the new generation of models can be used to look at large-scale (global) patterns of vegetation response to climate change. However, the resolution of these models currently is similar to that of the GCMs, and so they are not reliable at regional levels. Stand and biome-scale models also exist but are do not incorporate transient climate change effects; i.e. they assume an equilibrium between the vegetation and the climate (Waring and Running 1998).

ACKNOWLEDGMENTS

This project was funded by the Government of Canada's Climate Change Action Fund (Agreement A305), the Saskatchewan Research Council (SRC), Saskatchewan Environment and Resource Management (SERM), and the University of Saskatchewan. I thank Elaine Wheaton and Virginia Wittrock of the Saskatchewan Research Council for information on the Turc and Lecerf bioproductivity index; Jeff Thorpe of SRC for assisting in study site location and providing site data; Ted Hogg of the Canadian Forest Service for many useful discussions about impacts of climate change on moisture availability; Rory Mcintosh of Saskatchewan Environment and Resource Management for information on insect-climate relationships; Dan Lorer of SERM for providing forest inventory data; Mike Starr of the Finnish Forest Research Institute for advice on the use of the WATBAL water balance model; Glen Padbury and Marc St. Arnaud of the Saskatchewan Land Resource Unit for providing GIS analysis and soil survey data; and Marc-Andre Parisien of the CFS for providing the head-fire intensity data.

REFERENCES

Ayres, M.P. and M.J. Lombardero. 2000. Assessing the consequences of global change for forest disturbance from herbivores and pathogens. *The Science of the Total Environment*, **262**:263-286.

Blais, J.R. 1985. The ecology of the eastern spruce budworm: a review and discussion. *In*: Sanders, C.J., R.W. Stark, E.J. Mullins, and J. Murphy (Editors), *Recent Advances in Spruce Budworm Research*, pp. 49-59. Proceedings of the CANUSA Spruce Budworm Research Symposium, September 16 to 20, 1984, Bangor, Maine. Canadian Forestry Service, Ottawa, Ontario.

Camill, P. and J.S. Clark. 2000. Long-term perspectives on lagged ecosystem responses to climate change: permafrost in boreal peatlands and the grassland/woodland boundary. *Ecosystems*, **3**:534-544.

Carter, T.R., M. Hulme, and M. Lal. 1999. *Guidelines on the Use of Scenario Data for Climate Impact and Adaptation Assessment, Version 1.* Prepared by Intergovernmental Panel on Climate Change, Task Group on Scenarios for Climate Impact Assessment. Finnish Environment Institute, Helsinki, Finland.

Cash, D.W. 2000. Distributed assessment systems: an emerging paradigm of research, assessment and decision-making for environmental change. *Global Environmental Change*, **10**:241-244.

Caya, D. and R. Laprise. 1999. A semi-implicit lagrangian regional climate model: the Canadian RCM. *Monthly Weather Review*, **127**:341-362.

Clark, J.S.,E.C. Grimm, J. Lynch, and P.G. Mueller. 2001. Effects of Holocene climate change on the C4 grassland/woodland boundary in the northern plains, USA. *Ecology,* **82**:620–636.

Cramer, W., A. Bondeau, F.I. Woodward, I.C. Prentice, R.A. Betts, V. Brovkin, P.M. Cox, V. Fisher, J.A. Foley, A.D. Friend, C. Kucharik, M.R. Lomas, N. Kutty, S. Sitch, B. Smith, A. White, and C. Young-Molling. 2001.Global response of terrestrial ecosystem structure and function to CO_2 and climate change: results from six dynamic global vegetation models. *Global Change Biology*, **7**:357-373.

Flannigan, M., Y. Bergeron, O. Engelmark, and M. Wotton. 1998. Future wildfire in circumboreal forests in relation to global warming. *Journal of Vegetation Science*, **9**:469-476.

Flannigan, M.D., B. J. Stocks, and B.M. Wotton. 2000. Climate change and fires. *The Science of the Total Environment*, **262**:221-229.

Flannigan, M., I. Campbell, M. Wotton, C. Carcaillet, P. Richard, and Y. Bergeron. 2001. Future fire in Canada;'s boreal forest: paleoecology results and general circulation model - regional climate model simulations. *Canadian Journal of Forest Research,* **31**:854-864.

Fleming, R.A. 2000. Climate Change and Insect Disturbance Regimes in Canada's Boreal Forests. *World Resource Review*, **12**:521-555.

Forestry Canada Fire Danger Group. 1992. *Development and Structure of the Canadian Forest Fire Behavior Prediction System.* Information Report ST-X-3, Forestry Canada, Ottawa, ON.

Frank, Alan. *Personal Communication*, Prince Albert, SK, 27 March, 2001.

Hanson, P.J. and J.F. Weltzin. 2000. Drought disturbance from climate change: response of United States forests. *The Science of the Total Environment*, **262**:205-220.

Harris, L.D. 1984. *The Fragmented Forest: Island Biogeography Theory and the Preservation of Biotic Diversity*. University of Chicago Press, Chicago IL.

Herrington, R., B. Johnson, and F. Hunter, editors. 1997. *Responding to Climate Change in the Prairies. Volume III of the Canada Country Study: Climate Impacts and Adaptation*. Atmospheric Environment Branch, Environment Canada, Ottawa, Canada.

Hogg, E.H. 1994. Climate and the southern limit of the western Canadian boreal forest. Canadian *Journal of Forest Research*, **24**:1835-1845.

Hogg, E.H. and P. Hurdle. 1995. The aspen parkland in western Canada: A dry-climate analogue for the future boreal forest? *Water, Air and Soil Pollution*, **82**:391-400.

Hogg, E.H. and A.G. Schwarz. 1999. *Tree-ring analysis of declining aspen stands in west-central Saskatchewan.* Canadian Forest Service, Northern Forestry Centre, Information Report NOR-X-359. Edmonton, AB.

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

Johnston, M.H. 1996. Can harvesting emulate natural disturbance in Boreal Mixedwoods? Pp. 155-157 in: Smith, C.R. and G.W. Crook (compilers). *Advancing Boreal Mixedwood Management in Ontario: Proceedings of a Workshop.* Canadian Forest Service, Great Lakes Forestry Centre, Sault Ste. Marie, ON.

Kabzems, A., A.L. Kosowan, and W.C. Harris. 1986. *Mixedwood Section in an Ecological Perspective*. Technical Bulletin No. 8, Forestry Division, Saskatchewan Parks and Renewable Resources, Prince Albert, SK.

Kafka, V, M.-A. Parisien, K. Hirsch, B. and Todd. 2001. *Climate Change in the Prairie Provinces: Assessing Landscape Fire Behavior Potential and Evaluating Fuel Treatment as an Adaptation Strategy*. Final Report to Prairie Adaptation Research Collaborative, Regina, SK.

Kirschbaum, M., and A. Fischlin. 1996. Climate change impacts on forests. Pp. 95-129 in: Watson, R., M. Zinyowera, and R. Moss (Eds.). *Climate Change 1995: Impacts, Adaptations and Mitigation of Climate Change*. Cambridge University Press, Cambridge, UK.

Laprise, R., D. Caya, M. GiguPre, G. Bergeron, H. C^tJ, J.-P. Blanchet, G.J. Boer, and N.A. McFarlane. 1998. Climate and climate change in western Canada as simulated by the Canadian regional climate model. *Atmosphere-Ocean*, **36**:119-167.

Mattson, W.J. and R.A. Haack. 1987. The role of drought in outbreaks of plant-eating insects. *BioScience*, **37**:110-118.

Neilson, R.P. 1993. Transient ecotone response to climatic change: some conceptual and modeling approaches. *Ecological Applications*, **3**:385-395.

Neilson, R.P and S.W. Running. 1996. Global dynamic vegetation modeling: coupling biogeochemistry and biogeography models. Pp. 451-465 in Walker B. and W. Steffen (Eds.). *Global Change and Terrestrial Ecosystems. International Biosphere-Geosphere Programme Book Series, Volume 2*. Cambridge University Press, Cambridge UK.

Noble, I.R. 1993. A model of the responses of ecotones to climate change. *Ecological Applications*, **3**:396-403.

Noss, R.F., and B. Csuti. 1997. Habitat fragmentation. Pp. 269-304 in: Meffe, G.K. and R.C. Carroll (Eds.). *Principles of Conservation Biology, Second Edition*. Sinauer Associates, Sunderland, MA.

O'Shaughnessy, S. 2001. *A Framework for Assessing Climate Change Adaptation Options for the Forestry Sector in the Prairie Provinces*. M.Sc. Thesis, Department of Geography, University of Saskatchewan, Saskatoon, SK.

Peng, C. 2000. From static biogeographical model to dynamic global vegetation model: a global perspective on modelling vegetation dynamics. *Ecological Modelling*, **135**:33-54.

Pitelka, L.F. and the Plant Migration Workshop Group. 1997. Plant migration and climate change. *American Scientist*, **85**:464-73.

Roland, J. and P.D. Taylor. 1997. Insect parasitoid species respond to forest structure at different spatial scales. *Nature*, **386**:710-713.

Rothman, L. and J. Roland. 1998. Relationships between forest fragmentation and colony performance in the forest tent caterpillar, *Malacosoma disstria*. *Ecography*, **21**:383-391.

Shafer, S.L., P.J. Bartlein, and R.S. Thompson. 2001. Potential changes in the distributions of western North American tree and shrub taxa under future climate scenarios. *Ecosystems*, **4**:200-215.

Shields, J.A., C. Tarnocai, K.W.G. Valentine, and K.B. MacDonald. 1991. *Soil Landscapes of Canada: Procedures Manual and User's Handbook*. Agriculture Canada Publication 1868/E, Land Resource Research Centre, Agriculture Canada, Ottawa, ON.

Singh, T. and E. Wheaton. 1991. Boreal forest sensitivity to global warming: implications for forest management in western interior Canada. *The Forestry Chronicle*, **67**:342-348.

Solomon, A.M. and A.P. Kirilenko. 1997. Climate change and terrestrial biomass: what if trees do not migrate? *Global Ecology and Biogeography Letters*, **6**:139-148.

Starr, M. 1999. WATBAL: a model for estimating monthly water balance components, including soil water fluxes. In: Kleemola, S. and M. Forsius (Eds.). *Eighth Annual Report, U.N. E.C.E. Convention on Long-Range Transboundary Air Pollution, International Cooperative Programme on Integrated Monitoring of Air Pollution - Effects on Ecosystems*. The Finnish Environment 325:31-35. Finnish Environmental Institute, Helsinki, Finland.

Stocks, B.J. 1987. Fire potential in the spruce budworm-damaged forests of Ontario. *Forestry Chronicle*, **63**:8-14.

Stocks, B., M. Fosberg, T. Lynham, L. Mearns, M. Wotton, Q. Yang, J. Jin, K. Lawrence, G. Hartley, J. Mason, and D. McKenny. 1998. Climate change and forest fire potential in Russian and Canadian boreal forests. *Climate Change*, **38**:1-13.

Turc, L. and H. Lecerf. 1972. Indice climatique de potentialité agricole. *Science du Sol*, **2**:81-102.

Van Wagner, C.E. 1987. *Development and Structure of the Canadian Forest Fire Weather System*. Forestry Technical Report 35, Canadian Forest Service, Ottawa, ON.

Volney, W.J.A. and R.A. Fleming. 2000. Climate change and impacts of boreal forest insects. *Agriculture, Ecosystems and Environment*, **82**:283-294.

Waring, R. and S.W. Running. 1998. *Forest Ecosystems: Analysis at Multiple Scales*. Academic Press, San Diego, CA.

Watson, R., M. Zinyowera, and R. Moss (Eds.). 1996. *Climate Change 1995: Impacts, Adaptations and Mitigation of Climate Change*. Cambridge University Press, Cambridge, UK.

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

Weir, J.M.H. and E.A. Johnson. 1998. Effects of escaped settlement fires and logging on forest composition in the mixedwood boreal forest. *Canadian Journal of Forest Research*, **28**:459-467.

Wheaton, E.E. and V. Wittrock. 1994. Bioclimatic implications of global warming for the boreal and subarctic regions of western Canada. Pp. 9-14 in: Riewe, R. and J. Oakes (Eds.). *Biological Implications of Global Change: Northern Perspectives*. Occasional Publication 33, Canadian Circumpolar Institute, University of Alberta, Edmonton, AB.

Williams, G.D.V. 1985. Estimated bioresource sensitivity to climatic change in Alberta, Canada. *Climatic Change*, **7**:55-69.

Williams, G.D.V. and E.E. Wheaton. 1998. Estimating biomass and wind erosion impacts for several climate scenarios: a Saskatchewan case study. *Prairie Forum*, **23**:49-66.