Net primary productivity following forest fire for Canadian ecoregions

B.D. Amiro, J.M. Chen, and J. Liu

Abstract: Recent modelling results indicate that forest fires and other disturbances determine the magnitude of the Canadian forest carbon balance. The regeneration of post-fire vegetation is key to the recovery of net primary productivity (NPP) following fire. We geographically co-registered pixels classed using the Boreal Ecosystem Productivity Simulator, a process-based model with AVHRR (advanced very-high resolution radiometer) satellite estimates of leaf-area index and land cover type, with polygons from a recent database of large Canadian fires. NPP development with time since fire was derived for the first 15 years following the disturbance in the boreal and taiga ecozones. About 7×10^6 ha were analysed for over 500 fires occurring between 1980 and 1994. NPP increases linearly through this period, at rates that depend on ecoregion. A longer data set for the Boreal Plains ecozone of Alberta shows that NPP levels off at about 20–30 years and remains constant for 60 years. The NPP trajectories can be used as spatial averages to support models of forest carbon balance and succession through the most fire-prone regions of Canada.

Résumé : Les résultats de travaux récents de modélisation indiquent que les feux de forêt et autres perturbations jouent un rôle déterminant dans l'ampleur du bilan du carbone dans les forêts canadiennes. La régénération de la végétation qui vient après feu est l'élément clé dans la récupération de la productivité primaire nette suite à un feu. Nous avons co-enregistré géographiquement les pixels classifiés à l'aide du simulateur de productivité de l'écosystème boréal, un modèle basé sur les processus qui fonctionne à partir d'estimés satellitaires de l'indice de surface foliaire et du type de couvert, obtenus à l'aide d'un radiomètre à très haute résolution, avec des polygones provenant d'une base de données récente sur les feux de forêt majeurs au Canada. L'évolution dans le temps de la productivité primaire nette après feu a été établie pour les 15 premières années après une perturbation dans les écozones de la forêt boréale et de la taïga. Environ 7×10^6 ha ont été analysés pour plus de 500 feux survenus entre 1980 et 1994. La productivité primaire nette a augmenté de façon linéaire pendant cette période à des taux qui dépendent de la région écologique. Des données couvrant une plus longue période pour l'écozone des plaines boréales en Alberta montrent que le niveau de productivité primaire nette plafonne à environ 20–30 ans et reste constant pendant 60 ans. Les trajectoires de productivité primaire nette peuvent être utilisées comme des moyennes spatiales pour supporter les modèles de bilan du carbone et de succession des forêts à travers les régions du Canada les plus susceptibles d'être affectées par le feu.

[Traduit par la Rédaction]

Introduction

Fire is the main stand-renewing agent in the boreal forest of Canada (e.g., Weber and Flannigan 1997). On average, 1.3×10^6 ha of boreal forest have burned annually in Canada through most of this century, with extreme fire years having about five times this amount (Weber and Stocks 1998). The fire cycle (the time required to burn an area equal in size to the entire study area) typically ranges from 40 to 110 years through much of the boreal forest (e.g., Bergeron 1991; Larsen 1998), indicating the significant impact of fire on forest age. In the last few years, the issue of fire has become more important in discussions concerning the global carbon balance (e.g., Kasischke et al. 1995; Stocks et al. 1996). The carbon budget model of the Canadian forest sector suggests that disturbance (fire, insects) drives much of the net carbon balance (Kurz and Apps 1999). Forests provided a net sink for carbon through much of the early part of this century, but there has been a decrease in this sink through the 1970s and 1980s, such that Canadian forests may now be a net source of carbon (Kurz and Apps 1999) or a very small sink (J.M. Chen et al. 2000). Knowing the magnitude of the source–sink strength following fire is key to the overall terrestrial carbon cycle, even if much of the "unmanaged" forest is not explicitly part of the current Kyoto Protocol (IGBP 1998).

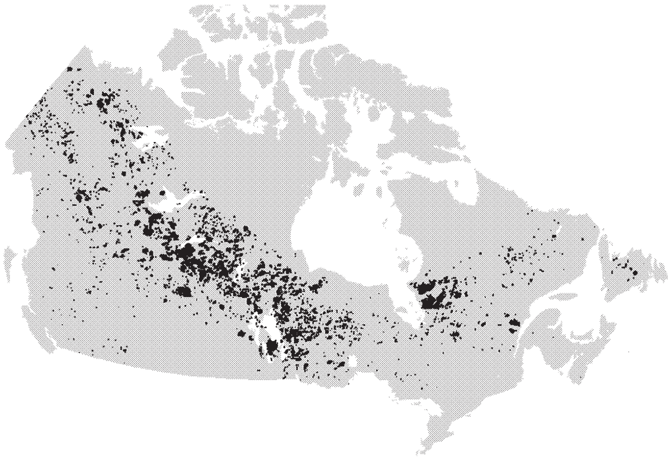
The accumulation of forest biomass drives the carbon balance in the early years following fire. Although some site-specific measurements are available (Dyrness and Norum 1983; Auclair 1985), it is difficult to develop successional trends that apply at large landscape spatial scales. A combination of satellite data and mathematical modelling has been used to derive large-scale estimates of net primary productivity (NPP) (Jang et al. 1996; Liu et al. 1997, 1999). This approach allows for spatial averaging, recognising that subpixel-scale variability can be large. NPP includes the net fixing of autotrophic carbon in above- and below-ground parts of the ecosystem. However, it does not provide the net carbon balance, which needs to include decomposition of woody debris and other soil carbon pools, as well as fire

Received October 25, 1999. Accepted February 7, 2000.

B.D. Amiro.[1] Canadian Forest Service, Northern Forestry Centre, 5320-122 Street, Edmonton, AB T6H 3S5, Canada.
J.M. Chen and J. Liu. Canada Centre for Remote Sensing, 588 Booth Street, Ottawa, ON K1A 0Y7, Canada.

[1]Corresponding author. e-mail: bamiro@nrcan.gc.ca

Fig. 1. Fires greater than 200 ha in area for the 1980–1993 period. The dark polygons represent actual areas burned.



effects on soil carbon. NPP is a good indicator of forest recovery following fire and is the primary driver for potential carbon sequestration.

A large-fire database is being developed, where all forest fires greater than 200 ha in area are explicitly mapped in a geographical information system (GIS). Figure 1 shows the actual fire polygons for the 1980–1993 period, with the greatest concentration of burned area extending throughout the boreal and taiga regions. These larger fires are most important, since only 2–3% of the fires account for 97–98% of the area burned during this period (Stocks 1991). This database includes spatial polygons of each fire, with supporting information on the month that the fire started, area burned, suppression actions, and cause of ignition. In some cases, large unburned islands are also mapped. To date, only the 1980–1998 data set is available for most of Canada, although earlier data are being included for many areas. This database presents an opportunity to investigate the development of NPP with time since fire on a large national scale. Many fires are large, singly covering more than 100 000 ha, which necessitates looking at the data at the ecozone or ecoregion scale. We have chosen Canadian ecoregions, or groups of ecoregions, as the primary spatial units to derive NPP trajectories with time since fire. This is a more natural grouping than provincial jurisdictions and can be easily investigated using the GIS-based ecoregion classifications (Lowe et al. 1996).

The scale of the NPP estimates are nominally 1 km^2, which is appropriate for giving a national perspective and is compatible with the scale of individual fires (i.e., >200 ha) within the large-fire database. Much larger pixels would smear landscape variability and only allow investigations of very large fires. Much smaller pixels would be helpful in estimating variability but would generate a larger data set. At this time, the limited period of the large-fire database only allows us to investigate the first 15 years of NPP development following fire for the whole country. This is a critical early period that influences the national forest carbon balance following years when large areas are burned. Our development of NPP trajectories is intended to provide additional evidence to be used in the carbon budget model of the Canadian forest sector (Kurz and Apps 1999) and in the integrated terrestrial ecosystem carbon budget model (W.J. Chen et al. 2000), as well as in other models of forest succession and impacts of fire on the forest (e.g., Peng and Apps 1999).

Methods

NPP model

A Canada-wide annual NPP map for 1994 is used in this study. It was produced using the boreal ecosystem productivity simulator model (BEPS), described by Liu et al. (1997, 1999). This model is modified from FOREST–BGC (Running and Coughlan 1988) and uses daily meteorological inputs to drive the process-based calculations of NPP. Spatially, the model uses advanced very-high resolution radiometer (AVHRR) satellite measurements to estimate leaf-area index (LAI) and land-cover type (Chen and Cihlar 1996; Cihlar et al. 1997) as model inputs. Soil water holding capacity was derived from Shields et al. (1991), and the meteorological inputs were based on gridded interpolations from climate stations. NPP is calculated on a nominal 1 × 1 km^2 pixel and has been derived for all of Canada (http://www.ccrs.nrcan.gc.ca/ccrs/tekrd/rd/apps/em/beps/bepse.html). Total NPP (i.e., above and below ground) is modelled as the gross primary productivity (GPP) minus the autotrophic respiration of all plant components (leaf, stem, root). The components of the model (radiation interception, evapotranspiration, rainfall interception, GPP, stomatal conductance, autotrophic respiration, etc.) were validated using tower flux and auxiliary data collected during the BOREAS study (Chen et al. 1999). Hourly and daily NPP calculations using BEPS were also validated using CO_2 flux measurements above and below the canopy. The accuracy of the NPP calculation for the mean area value of correctly classified pixels is estimated to be 60% for single pixels and 75% for a nine-pixel area, based on a limited amount of ground sampling (Liu et al. 1997). However, the model has not been validated for all of Canada.

Ideally, the response of a forest ecosystem after fire should be studied by following the NPP sequence in consecutive years after the disturbance. However, this would require multiple-year NPP maps, but we only have a map available for 1994 because of the huge effort required to produce such a map. An alternative approach, which we have followed, is to study differences in NPP during a single year for fire events that occurred during a variety of previous years. We recognise that these two approaches will yield different results depending on the amount of climatic variability among years.

Post-fire NPP estimates

We combined the large-fire database with the NPP map of Canada to look at the NPP values following fires of a given year. This was done by overlaying the fire-boundary polygon onto the NPP map within the GIS. The data set has NPP estimated in increments of 5 g C·m^{-2}·year^{-1}, and we classed the values into ranges of 50 g C·m^{-2}·year^{-1}, which were then colour coded. Pixels with NPP values of <5 g C·m^{-2}·year^{-1} were invariably classified as waterbodies. Therefore, the minimum possible value for a land surface was 27.5 g C·m^{-2}·year^{-1} (i.e., the mean value for the 5–50 g C·m^{-2}·year class^{-1}).

The pixels of a given class were counted manually within each fire, and we excluded pixels at the edges of fire boundaries and lakes. Although tedious, this manual count provided more control than a programmed automated procedure. It also allowed for some adjustment where fire-polygon boundaries did not exactly coincide with NPP map pixels (<1%). Only the most recent burn age was used if fire polygons from different years coincided. This rarely happened for most ecoregions since overburning was rare. However,

Fig. 2. Map of ecoregions analysed. The numbers identify ecoregions as described in the text based on Lowe et al. (1996). The four ecozones are shown as different hatched areas.

there were several areas where more than one fire occurred in the longer term data set for Alberta.

The area examined was grouped by ecoregion (Lowe et al. 1996), but some ecoregions had few fires or were of a small area. Eight individual ecoregions had a sufficient number of fires in the database to be included, and two other groupings were used by amalgamating ecoregions that had some common features. Where large fires traversed ecoregion boundaries, the fire was split into the respective ecoregions. These ecoregions were part of either the boreal or taiga groups of ecozones. For most ecoregions, the 1980–1993 period was analysed (1994 was excluded), but we also used the province of Alberta digital database dating back to 1931. Regression analyses were performed using SYSTAT (SPSS Inc. 1997).

Results

Figure 2 shows the location of the ecoregions. Within the Boreal Shield ecozone, ecoregions 87 (Athabasca Plain), 88 (Churchill River Upland), 89 (Hayes River Upland), 90 (Lac Seul Upland), and 95 (Big Trout Lake) were analysed. Within the Taiga Shield ecozone, ecoregions 69 (Tazin Lake Upland), 71 (Selwyn Lake Upland), and 72 (La Grande Hills) were analysed. All of the ecoregions within the Taiga Plains ecozone were grouped together because of the relatively small size of each ecoregion, and the need to get a sufficient number of fires in an area. Similarly, ecoregions 139–141, 147, and 150–154 were grouped as the Mid-boreal Uplands ecoregion grouping within the Boreal Plains ecozone. The Boreal Plains ecozone within Alberta was analysed separately as a longer term database.

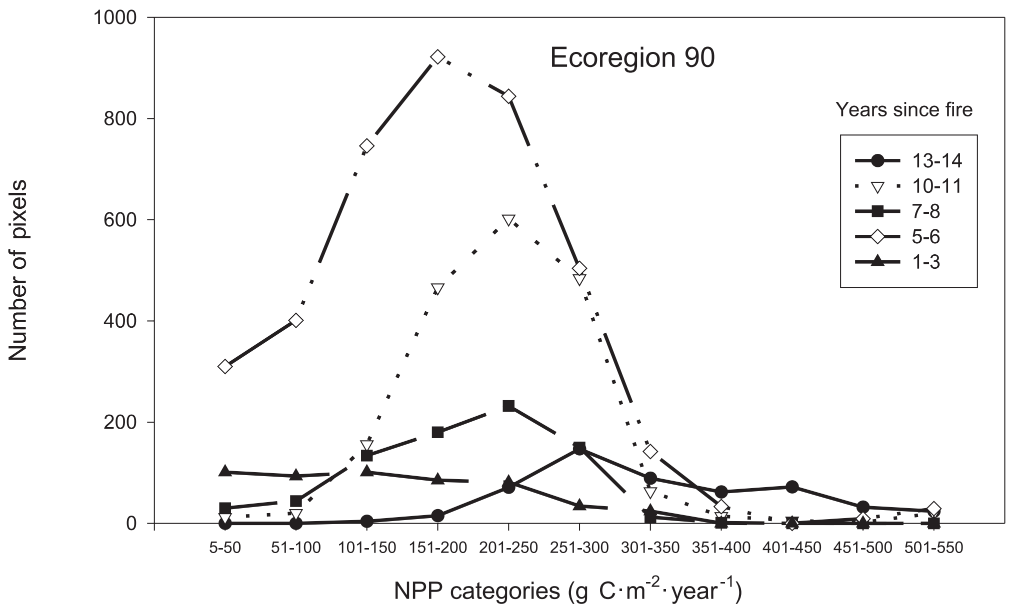
For the 1980 to 1993 period, we analysed 502 fires within the 10 ecoregion groups. We only looked at the larger fires, where between 9 and 1988 pixels (about 1000 – 200 000 ha) were counted within individual fires, with a mean of 145 pixels per fire. For the Alberta 1931–1994 data set, 181 fires were analysed covering between 10 and 1130 pixels, with a mean of 114. Pixels were grouped for a given ecoregion and given year of burn (i.e., all fires were grouped). This provided a spatial analysis that did not bias towards smaller fires. Table 1 gives the number of pixels analysed, showing the uneven data set, especially for years when there were no fires in a given ecoregion. A total of 73 427 pixels were analysed, representing about 7×10^6 ha in the 10 ecoregions.

Each ecoregion and year had a distribution of pixels where the peak of the distribution shifted to progressively higher NPP values with time since fire. An example of 2-year fire groupings for ecoregion 90 is shown in Fig. 3. Absolute values of the number of pixels are plotted to separate the distributions visually. Only the most recent fires (1–3 years since fire) have the mode located in the lower NPP classes.

Table 1. Number of pixels for each ecoregion.

Fire year	Ecoregion 87	88	89	90	95	139	69	71	72	Taiga Plains
1980	2229	2383	374	387	78	1647	1512	574	0	1873
1981	340	7584	48	129	409	3670	145	114	236	3455
1982	0	162	0	0	0	684	13	40	0	1286
1983	35	0	12	1295	13	0	53	0	0	432
1984	615	320	154	549	0	188	0	130	0	174
1985	90	140	0	0	0	15	62	589	257	0
1986	0	0	171	506	0	0	0	0	0	947
1987	0	444	544	277	17	394	0	0	0	1858
1988	67	0	481	2078	37	194	0	188	990	277
1989	295	4032	3223	1862	340	84	77	2223	7098	1302
1990	59	362	26	0	0	288	0	0	0	157
1991	0	53	176	423	560	0	100	898	35	42
1992	310	0	0	0	449	0	0	1163	0	0
1993	311	1071	0	96	0	0	0	48	0	2299

Fig. 3. Distribution of pixels in NPP categories for fires in ecoregion 90 (see Fig. 2).



We tested the effect of the relatively broad 50 g C·m^{-2}·year^{-1} NPP categories on estimating mean NPP for an individual fire by dividing the lowest 5–50 g C·m^{-2}·year^{-1} category into divisions of 5 g C·m^{-2}·year^{-1}. For ecoregion 90, there were seven fires in the 1 to 3 years-since-fire period, two of which had no pixels and two of which had only four pixels in the 5 to 50 g C·m^{-2}·year^{-1} category. The other three fires had from 23 to 35% of their pixels in this category (15–61 pixels), and the mean NPP values for these ranged from 25 to 35 g C·m^{-2}·year^{-1}. This compares favourably with our assumption that the lowest category has a mean value of 27.5 g C·m^{-2}·year^{-1}, and it is likely that our NPP categories estimate the underlying distribution within about 10 g C·m^{-2}·year^{-1}.

Figure 4 shows the NPP relationship with time since fire for six ecoregions in the boreal ecozones. Mean values ± 1 SD are given for all fires in each year. The trend appears to be approximately linear, and we have calculated linear regression lines based on the mean values for each year. This weights the regression with the time since fire, as opposed to using a regression of all data that would weight the regression to the years with a large number of pixels. Upward trends are shown for all ecoregions, except for the Mid-boreal Uplands, where there is little change with time since fire. Figure 5 shows the NPP relationship for the taiga ecoregions. These values are much lower than the boreal ecoregions (note different ordinate scale), and all appear to be increasing over time, with a slower increase for ecoregion 71. The regression equations and r^2 values are presented on the figures for each ecoregion. Slopes greater than about 7 g C·m^{-2}·year^{-1} per year since fire are significantly different from zero so that the regression slopes for ecoregions 71, 72, 89, and Mid-boreal Uplands are not significant at the P = 0.05 level. All significant slopes are statistically different

Fig. 4. NPP for the boreal ecoregions. The points show the mean value of all pixels for a given year (*n* given in Table 1), and the error bars are ±1 SD. The regression lines are based on the mean values, and the equations use the units shown on the axes.

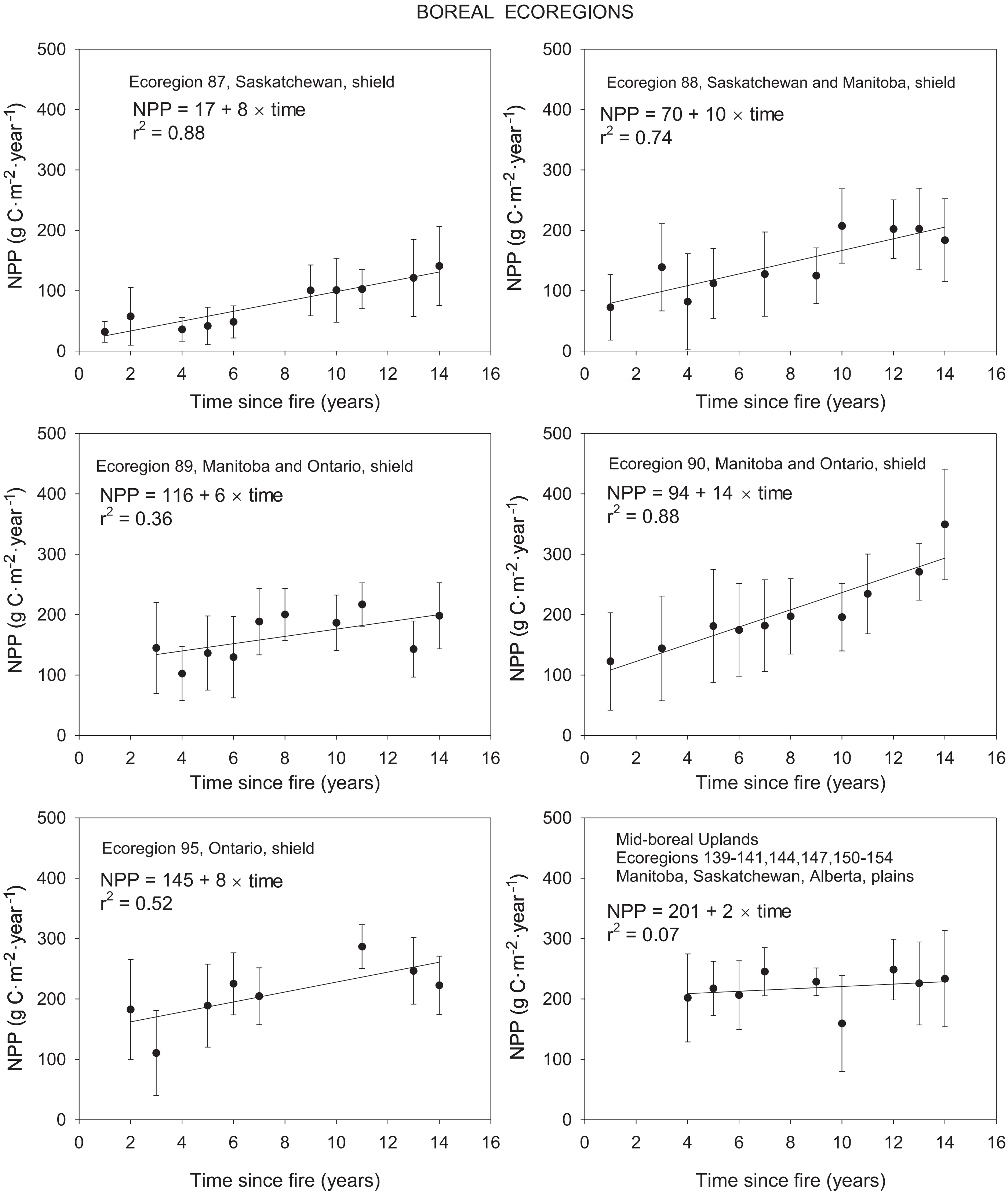
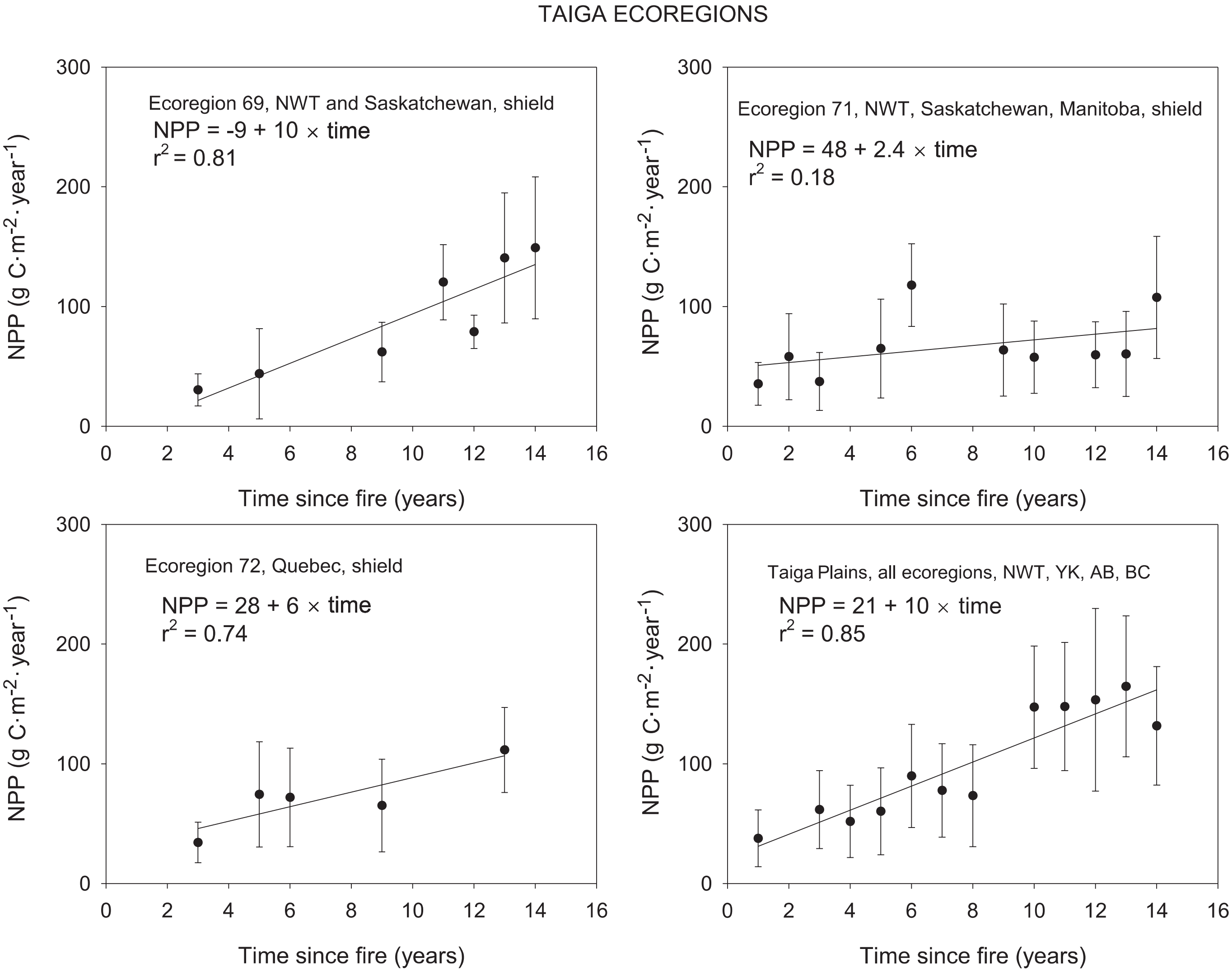


Fig. 5. NPP for the taiga ecoregions. The points show the mean value of all pixels for a given year (n given in Table 1), and the error bars are ±1 SD. The regression lines are based on the mean values, and the equations use the units shown on the axes.



from each other ($P < 0.05$; SYSTAT, general linear models procedure), with the exception that the slopes for ecoregions 69 and 72 are the same ($P = 0.75$). The similar slopes for these two ecoregions is likely caused by the smaller data set with fire only occurred in 5 years in ecoregion 72.

We also did regressions using each individual fire, instead of all pixels for a given year. This scheme gives equal weighting to all fires, as opposed to equally weighting the landscape. However, it tends to have little effect on the results. For example, a regression of all fires for ecoregion 90 gives a slope of 13 g $C \cdot m^{-2} \cdot year^{-1}$ per year since fire, compared with a slope of 14 g $C \cdot m^{-2} \cdot year^{-1}$ per year since fire using the mean of all pixels.

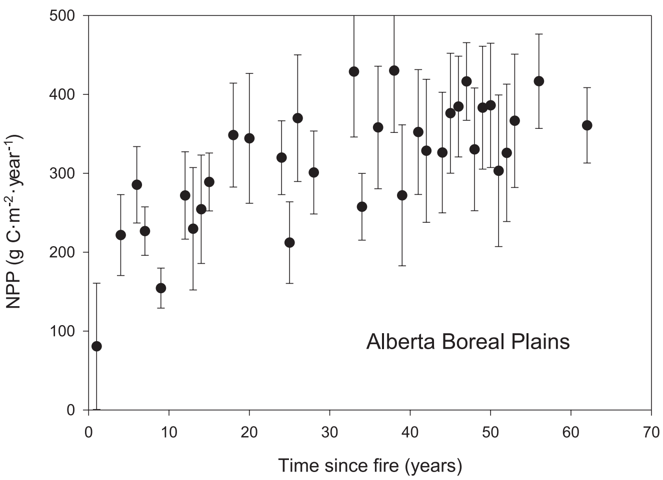
A longer time frame is available for the Boreal Plains area of Alberta (Fig. 6). It appears that NPP increases for about the first 20 years following fire and then remains at about a constant value. For the period of 20–62 years post-burn, the mean (±1 SD) NPP value is 366 ± 88 g $C \cdot m^{-2} \cdot year^{-1}$ for 1994. The mean value for the 30- to 62-year period is identical, and a regression of the 16 fires with time since fire for the year 20–30 period has an insignificant slope ($P = 0.78$). This suggests that at least for some ecoregions, the approximately linear increase shown in the early years in Figs. 4 and 5 likely reaches a steady value sometime after 20 years. However, the period is likely even longer in many areas with poorer growing conditions.

Discussion

Reliability of NPP estimates

The validation of the modelling technique is described by Liu et al. (1997, 1999) and Chen et al. (1999). NPP is estimated using satellite imagery, weather station data, and mathematical modelling. This combination of data helps to decrease the reliance on a single data source. However, it also means that errors and uncertainty are introduced from each data source. Our goal is to compare NPP estimates with time since fire for a given ecoregion. The weather station data are relatively similar for a given ecoregion and probably do not cause much error in the relative estimates for the single year of the model output. For a single ecoregion, most of the differences among sites will be driven by differential

Fig. 6. NPP for Alberta Boreal Plains, 1931–1993 period. The points show the mean value of all pixels for a given year, and the error bars are ±1 SD.



satellite estimates of reflectance, inferring the amount of vegetation present. Therefore, the largest source of error is likely caused by the ability of the AVHRR system to correctly estimate meaningful pixel averages of the surface vegetation. Liu et al. (1997) discuss the estimation of LAI and land-cover types from the AVHRR data. However, we have not validated cases using field data where vegetation is sparse, as is the case in the early years following fire.

Our nominal 1-km^2 pixel average could include a wide variety of subpixel features such as unburned islands, bare rock, and blackened vegetation and soil. This gives us a "landscape" view of changes, but stand-level changes are more variable. For fires greater than 200 ha in area, Eberhart and Woodard (1987) found that about 4–5% of the fire area in northern Alberta consisted of unburned islands, with a median island size of 9 ha for fires greater than 2000 ha. We would expect then that many of these islands are included within pixels, but some of the larger ones can be greater than 1 km^2.

Early successional species including herbs and grasses may cover some areas quickly, whereas our model usually assumed that a coniferous forest land-cover type was present. We tested the potential error caused by incorrectly assuming a certain land-cover type. We ran BEPS for a single climate data set near Thompson, Man., over a range of LAI values from 1 to 5 m^2 leaf/m^2 for land-cover types of grass, coniferous forest, and deciduous forest. The coniferous forest and the grass underestimated NPP for the deciduous forest type by about 30–35%, whereas the coniferous forest was only about 5% less than the grass. Although we don't have a true "successional" land-cover type, this suggests that our potential error in the early post-fire years is less than 35%. We may underestimate early successional NPP in some ecoregions, but this may not be the case for the Mid-boreal Uplands, where a deciduous forest predominates and NPP is quite high, even 4 years following fire (Fig. 4).

Spatial autocorrelation is an important feature of fires on the landscape. This is because fire is a contagion process that is affected by weather and fuel. Hence, NPP development following fire has inherent spatial autocorrelation caused by fire severity, microenvironment, topography, seed sources, and other landscape factors. These spatial relationships were not explored here because of the complexity of factors that need to be considered to make meaningful conclusions.

Chronosequences of NPP

The chronosequences of NPP shown in Figs. 4–6 are for 1994 only. Therefore, they do not reflect NPP development throughout the full post-fire period for any given location. Also, analysis for a different year could show different absolute values, mostly because of different climate conditions even if LAI was constant. Running and Coughlan (1988) compare different climates for a given value of LAI showing about 50% variability in temperate climates across the continental United States. In our study, the different ecoregions had different climates that are included in the analysis, but it is important to note that only a single year of climate data is used. The regressions could change if multiple years of climate data are considered. In most cases, this will affect the whole ecoregion, such that all sites would experience different NPP values. Although the response to weather is not linear, much of the change would likely occur in the same direction, so that the relative NPP values may be similar to those shown in Figs. 4–6. One advantage of using the single-year NPP values is that post-fire growth dynamics are more closely related to physiological response under a single climate, thereby minimizing the effect of a varying climate.

The large-fire database and the NPP model are completely independent. Therefore, the NPP model used no prior knowledge of fire history, and successional trends with time are not contaminated by model inputs. Our use of fires >200 ha weights the NPP trends to conditions following stand-replacing fires. These larger fires would typically have burned for many days and include a large range of fire-behaviour characteristics. However, in many cases, the large fire runs occur when conditions are extreme with high-intensity crown fires burning large areas in a short time. In surface-fire conditions, vegetation would not be affected to the same extent, and actual NPP may not show the increasing trend with time since fire.

The regression slopes are significant for six of the 10 ecoregions. The regressions become significant when the slope exceeds about 7 g C·m^{-2}·year^{-1} per year since fire. This is similar to the variability, since the maximum standard error among years ranges from 3 to 11 g C·m^{-2}·year^{-1} among the ecoregions. Although not statistically significant, upward trends are still evident for ecoregions 72 and 89 (Figs. 4 and 5). The reason for the lack of trend in ecoregion 71 and the Mid-boreal Uplands is not obvious. These ecoregions are very different and our mean NPP estimates for all years differ by more than a factor of three between these two ecoregions (ecoregion 71, 66 g C·m^{-2}·year^{-1}; Mid-boreal Uplands, 218 g C·m^{-2}·year^{-1}).

With the exception of ecoregion 69, all of the regressions show positive intercepts. Although the data should not be extrapolated, good growth is often observed in the same year as the fire and definitely in the first post-fire year (e.g., Ohmann and Grigal 1979; Auclair 1985). The start of regeneration depends on the season of the fire, and we did not consider this in our analysis, thereby excluding areas burned in 1994. Regeneration studies show that most tree seedlings become established within 3–10 years following fire (St-Pierre et al. 1992; Lavoie and Sirois 1998; Greene and Johnson 1999). However, much of the vegetation contributing to LAI in these early years is shrub and ground vegetation.

NPP values for mature boreal forest communities are of the order of 200–500 g C·m^{-2}·year^{-1} (Gower et al. 1997). Following a fire in Minnesota, aboveground NPP was about 60 g C·m^{-2}·year^{-1} in the first year with about 150 g C·m^{-2}·year^{-1} in each of the next 3 years (Ohmann and Grigal 1979). Auclair (1985) measured an average of 126 g biomass·m^{-2}·year^{-1} (i.e., 63 g C·m^{-2}·year^{-1}) in northern Quebec lichen woodlands from year 1 to 15 after fire, with about 140 g C·m^{-2}·year^{-1} in the first year (including below ground). Additionally, Dyrness and Norum (1983) measured from 18 to 29 g C·m^{-2}·year^{-1} above ground in the first 3 years following fires in Alaska. In most of these studies, NPP increases with time in the early years. Our estimates are in the same general range as these site-specific studies, considering the large belowground biomass that was not measured in some studies as well as the large range in NPP among studies. Unlike the site-specific studies, our combination of the NPP model and large-fire database allows for large spatial estimates, more directly applicable to regional carbon cycle estimates.

Steady-state values for NPP were not reached within 15 years after fire for most of the ecoregions. The longer data set for the Alberta Boreal Plains suggests that NPP becomes relatively constant for a period of 20–60 years following fire. However, this steady-state period is dominated by tree canopy closure, with a constant value of LAI for many of the forest communities. This situation could take a much longer period to develop in the more northerly areas, such as the taiga ecozone. Without a longer data set in these areas, it is difficult to predict when steady state would be reached, and when a downward trend occurs because of stand degradation. Pare and Bergeron (1995) measured a biomass increase from year 27 to 75 following fire in Quebec with a decrease in subsequent years caused by stand degradation, implying a similar trend in NPP. In a landscape survey of Siberia, the maximum NPP was found at an age of 50–60 years (Gower et al. 1996). Our data suggest that biomass is still accumulating at 60 years post-fire in the Alberta Boreal Plains. If forest stands were beginning to degrade, it is likely that the satellite measurement of LAI would reflect this.

Other disturbances, such as insects, disease, and harvesting, also affect NPP. Harvesting is relatively unimportant for most of the ecoregions studied, since salvage logging became more widely practised only during the 1990s. Therefore, our estimates of NPP development following fire do not include salvage-harvesting impacts, and we have not separately looked at NPP following harvesting. Insects and disease are implicit agents throughout the forest, and we have not analysed for their impacts.

Direct measurements of net carbon dioxide flux along a 500-km transect over the boreal forest indicate that it takes between 15 and 30 years after fire to reach the same photosynthetic rates as more mature areas (Amiro et al. 1999). This period of a few decades for carbon flux to recover from fire is also often assumed in carbon balance models (Kasischke et al. 1995; Kurz and Apps 1999). However, carbon-balance models need data on net ecosystem productivity (NEP), which includes decomposition processes and non-autotrophic soil carbon dynamics. These are not part of our NPP estimates, since we have not included the decay of coarse woody debris that remains following fire. Similarly, we cannot detect changes in soil carbon caused by post-fire effects on respiration (Weber 1985, 1990; Pietikainen and Fritze 1993). Therefore, our NPP values remain as only part of the net carbon balance when the effect of fire on boreal forest carbon sink strength is estimated.

Conclusions

Our NPP trajectories with time since fire show approximately linear increases for most ecoregions for about the first 15 years post-disturbance. It is likely that steady state is not reached until at least 20 years, as shown by the longer trajectories for the Alberta Boreal Plains ecozone. Trajectories for longer periods in the ecoregions may not be easily extended in many cases, because some of the older mapped fire polygons are not exact. Hence, our more recent data are the most reliable. The regression equations were developed for the climate of 1994 only, but the relative magnitude of NPP development over time is likely similar during other years, although the absolute value may be different.

The NPP results provide a major component of the net carbon balance that can be used in carbon budget models for the Canadian boreal forest. The data are spatially extensive, covering about 7×10^6 ha, through the most fire-prone areas. We recommend that these estimates be used as part of the evidence of the effect of disturbance on the Canadian and global forest carbon balance.

Acknowledgements

We acknowledge the huge effort by the many individuals who contributed to the large-fire database from provincial, territorial, and federal agencies. We thank Ron Hall for help with data conversions and David Price for comments on an early draft of the manuscript.

References

Amiro, B.D., MacPherson, J.I., and Desjardins, R.L. 1999. BOREAS flight measurements of forest-fire effects on carbon dioxide and energy fluxes. Agric. For. Meteorol. **96**: 199–208.

Auclair, A.N.D. 1985. Postfire regeneration of plant and soil organic pools in a *Picea mariana – Cladonia stellaris* ecosystem. Can. J. For. Res. **15**: 279–291.

Bergeron, Y. 1991. The influence of island and mainland lakeshore landscapes on boreal forest fire regimes. Ecology, **72**: 1980–1992.

Chen, J.M., and Cihlar, J. 1996. Retrieving leaf area index of boreal conifer forests using LANDSAT TM images. Remote Sens. Environ. **55**: 153–162.

Chen, J.M., Liu, J., Cihlar, J., and Goulden, M.L. 1999. Daily canopy photosynthesis model through temporal and spatial scaling for remote sensing applications. Ecol. Modell. **124**: 99–119.

Chen, J.M., Chen, W.J., Liu, J., Cihlar, J., and Gray, S. 2000. Annual carbon balance of Canada's forests during 1895–1996. Global Biogeochem. Cycles. In press.

Chen, W.J., Chen, J.M., Liu, J., and Cihlar, J. 2000. Approaches for reducing uncertainties in regional forest carbon balance. Global Biogeochem. Cycles. In press.

Cihlar, J., Beaubien, J., Xiao, Q., Chen, J., and Li, Z. 1997. Land cover of the BOREAS region from AVHRR and LANDSAT data. Can. J. Remote Sens. **23**: 164–175.

Dyrness, C.T., and Norum, R.A. 1983. The effects of experimental fires on black spruce forest floors in interior Alaska. Can. J. For. Res. **13**: 879–893.

Eberhart, K.E., and Woodard, P.M. 1987. Distribution of residual vegetation associated with large fires in Alberta. Can. J. For. Res. **17**: 1207–1212.

Gower, S.T., McGuirie, R.E., and Murty, D. 1996. Aboveground net primary production decline with stand age: potential causes. Trends Ecol. Evol. **11**: 378–382.

Gower, S.T., Vogel, J.G., Norman, J.M., Kucharik, C.J., Steele, S.J., and Stow, T.K. 1997. Carbon distribution and aboveground net primary production in aspen, jack pine, and black spruce stands in Saskatchewan and Manitoba, Canada. J. Geophys. Res. **102**: 29 029 – 29 041.

Greene, D.F., and Johnson, E.A. 1999. Modelling recruitment of *Populus tremuloides*, *Pinus banksiana*, and *Picea mariana* following fire in the mixedwood boreal forest. Can. J. For. Res. **29**: 462–473.

International Geosphere–Biosphere Programme Terrestrial Carbon Working Group (IGBP). 1998. The terrestrial carbon cycle: implications for the Kyoto protocol. Science (Washington, D.C.), **280**: 1393–1394.

Jang, C.J., Nishigami, Y., and Yanagisawa, Y. 1996. Assessment of global forest change between 1986 and 1993 using satellite-derived terrestrial net primary productivity. Environ. Conserv. **23**: 315–321.

Kasischke, E.S., Christensen, N.L., Jr., and Stocks, B.J. 1995. Fire, global warming, and the carbon balance of boreal forests. Ecol. Appl. **5**: 437–451.

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.

Larsen, C.P. 1998. Spatial and temporal variations in boreal forest fire frequency in northern Alberta. J. Biogeogr. **24**: 663–673.

Lavoie, L., and Sirois, L. 1998. Vegetation changes caused by recent fires in the northern boreal forest of eastern Canada. J. Veg. Sci. **9**: 483–492.

Liu, J., Chen, J.M., Cihlar, J., and Park, W.M. 1997. A process-based boreal ecosystem productivity simulator using remote sensing inputs. Remote Sens. Environ. **62**: 158–175.

Liu, J., Chen, J.M., Cihlar, J., and Chen, W. 1999. Net primary productivity distribution in the BOREAS study region from a process model driven by satellite and surface data. J. Geophys. Res. **104**: 27 735 – 27 754.

Lowe, J.J., Power, K., and Marsan, M.W. 1996. Canada's forest inventory 1991: summary by terrestrial ecozones and ecoregions. Can. For Serv. Pac. For. Cent. Inf. Rep. BC-X-364E.

Ohmann, L.F., and Grigal, D.F. 1979. Early revegetation and nutrient dynamics following the 1971 Little Sioux Forest Fire in northeastern Minnesota. For. Sci. Monogr. **21**: 1–80.

Pare, D., and Bergeron, Y. 1995. Above-ground biomass accumulation along a 230-year chronosequence in the southern portion of the Canadian boreal forest. J. Ecol. **83**: 1001–1007.

Peng, C., and Apps, M.J. 1999. Modelling the response of net primary productivity (NPP) of boreal forest ecosystems to changes in climate and fire disturbance regimes. Ecol. Modell. **122**: 175–193.

Pietikainen, J., and Fritze, H. 1993. Microbial biomass and activity in the humus layer following burning: short-term effects of two different fires. Can. J. For. Res. **23**: 1275–1285.

Running, S.W., and Coughlan, J.C. 1988. A general model of forest ecosystem processes for regional applications. I. Hydrological balance, canopy exchange and primary production processes. Ecol. Modell. **42**: 125–154.

Shields, J.A., Tarnocai, C., Valentine, K.W.G., and Macdonald, K.B. 1991. Soil landscapes of Canada: procedures manual and user's handbook. Agric. Can. Publ. 1868/E.

Stocks, B.J. 1991. The extent and impact of forest fires in northern circumpolar countries. *In* Global biomass burning: atmospheric, climatic and biospheric implications. *Edited by* J.S. Levine. MIT Press, Cambridge, Mass. pp. 197–202.

Stocks, B.J., Lee, B.S., and Martell, D.L. 1996. Some potential carbon budget implications of fire management in the boreal forest. *In* Forest ecosystems, forest management and the global carbon cycle. *Edited by* M.J. Apps and D.T. Price. NATO ASI Ser. 40. pp. 89–96.

St-Pierre, H., Gagnon, R., and Bellefleur, P. 1992. Régénération après feu de l'épinette noire (*Picea mariana*) et du pin gris (*Pinus banksiana*) dans la forêt boréale, Québec. Can. J. For. Res. **22**: 474–481.

SPSS Inc. 1997. SYSTAT software, version 7.0 edition. SPSS Inc., Chicago, Ill.

Weber, M.G. 1985. Forest soil respiration in eastern Ontario jack pine ecosystems. Can. J. For. Res. **15**: 1069–1073.

Weber, M.G. 1990. Forest soil respiration after cutting and burning in immature aspen ecosystems. For. Ecol. Manage. **31**: 1–14.

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.

Weber, M.G., and Stocks, B.J. 1998. Forest fires and sustainability in the boreal forests of Canada. Ambio, **27**: 545–550.