Third Millennium Forestry: What climate change might mean to forests and forest management in Ontario

by William C. Parker[1,2], Stephen J. Colombo[2], Marilyn L. Cherry[2], Michael D. Flannigan[3], Sylvia Greifenhagen[2], Robert S. McAlpine[4], Chris Papadopol[2], and Taylor Scarr[4]

Climate change may profoundly influence Ontario's forest ecosystems and their management. Elevated atmospheric CO_2 concentrations, increased temperature and altered precipitation regimes will affect forest vegetation through their influence on physiological (e.g., photosynthesis, respiration) and ecological processes (e.g., net primary production, decomposition), and may result in dramatic northward shifts in the natural range of forest types and species. More importantly, climate change is expected to increase the frequency of natural disturbances. Silvicultural intervention will increasingly be relied on to maintain forest health, manage declining stands, regenerate disturbed areas and cutovers with desired species and genotypes, maintain genetic diversity, and assist in species migration. Given the increasingly important role of Ontario's forests in national and provincial efforts to meet greenhouse gas emission reduction targets of the Kyoto Protocol, afforestation, conservation of existing forests, and increased forest management activities to accelerate the storage of carbon in Ontario's forests will be key aspects of forestry at the start of the third millennium.

Key words: adaptation, afforestation, bioenergy, carbon dioxide, climate change, disturbance, intensive forest management, migration, mitigation, sequestration, succession

Le changement climatique pourrait modifier en profondeur les écosystèmes forestiers de l'Ontario et la façon de les aménager. Les concentrations élevées de CO_2 atmosphérique, la température plus élevée et les régimes de précipitations modifiées affecteront la végétation forestière suite à leurs influences sur les processus physiologiques (p. ex. la photosynthèse, la respiration) et écologiques (p. ex. la production nette primaire, la décomposition), et pourraient entraîner des déplacements dramatiques vers le nord de la distribution naturelle des types de forêts et des espèces. De façon encore plus importante, on s'attend à ce que le changement climatique accroisse la fréquence des perturbations naturelles. Les interventions sylvicoles dépendront de plus en plus du maintien de la santé des forêts, de l'aménagement des peuplements en déclin, de la régénération des superficies perturbées et coupées au moyen d'espèces et de génotypes désirés, du maintien de la diversité génétique, et de l'aide dans la migration des espèces. Étant donné le rôle de plus en plus importants que jouent les forêts de l'Ontario au niveau des efforts nationaux et provinciaux de correspondance aux objectifs de réduction des gaz à effets de serre selon le protocole de Kyoto, le boisement, la conservation des forêts actuelles, et l'accroissement des activités d'aménagement forestier pour accélérer la séquestration dur carbone dans les forêts ontariennes seront des aspects dominants en foresterie au début de troisième millénaire.

Mots-clés: adaptation, boisement, bioénergie, dioxyde de carbone, changement climatique, perturbation, aménagement forestier intensif, migration, mitigation, séquestration, succession

Introduction

Since the pre-industrial era, combustion of fossil fuels, land use changes, and deforestation have increased the CO_2 concentration of the earth's atmosphere from about 270 ppmv (parts per million by volume) to its present level of about 360 ppmv. At the current rate of increase of 1–2 ppmv per year (Keeling *et al.* 1995), atmospheric CO_2 concentration will reach 650–700 ppmv by 2075 (Houghton *et al.* 1995). Elevated CO_2 and increasing levels of other "greenhouse gases" (e.g., methane, nitrous oxide, chlorofluorocarbons) from biogenic and industrial sources (Mooney *et al.* 1987, Kreileman and Bouwman 1994) are expected to alter the global radiative energy balance and change the earth's climate at an unprecedented rate.

The Intergovernmental Panel on Climate Change (IPCC 1996) projects that a doubling of atmospheric CO_2 in the next century will increase average global surface air temperatures by 1 to 3.5°C, making this the warmest time of the present interglacial period. This temperature increase will vary spatially, with the largest changes at high latitudes. Warming is also predicted to be greater in winter than summer and at nighttime compared to daytime. Higher average temperatures will be accompanied by global increases in evaporation, cloudiness, and precipitation, but the amount and distribution of precipitation will vary regionally, with some areas having a higher frequency and severity of droughts (IPCC 1996). A changing climate may also increase the occurrence of extreme weather events, such as thunderstorms and floods (Francis and Hengeveld 1998).

If forecast changes in climate are realized, trees that begin growing in the next decade will mature in a climate substantially different from today. If trees and other forest organisms are ill adapted to future climatic conditions, foresters may require innovative silvicultural treatments to maintain productive ecosystems. Further, forest management to increase carbon (C) storage could become a critical component of national efforts to reduce greenhouse gas emissions and slow the rate of climate change. Clearly, the timely development and selective use of innovative forest management approaches in response to climate change are imperative given the inherently long planning scale of forestry and the ecological, social, and economic importance of forests to Canada.

The widely varying ecoclimatic zones in Canada will require regional strategies of forest adaptation and mitigation

[1]Author to whom correspondence should be directed. E-mail: bill.parker@mnr.gov.on.ca
[2]Ontario Forest Research Institute, 1235 Queen St. East, Sault Ste. Marie, ON P6A 2E5.
[3]Canadian Forest Service, Northern Forestry Centre, 5320 – 122nd St., Edmonton, AB T6H 3SH.
[4]Ontario Ministry of Natural Resources, Suite 400, Roberta Bondar Place, 70 Foster Dr., Sault Ste. Marie, ON P6A 6V5.

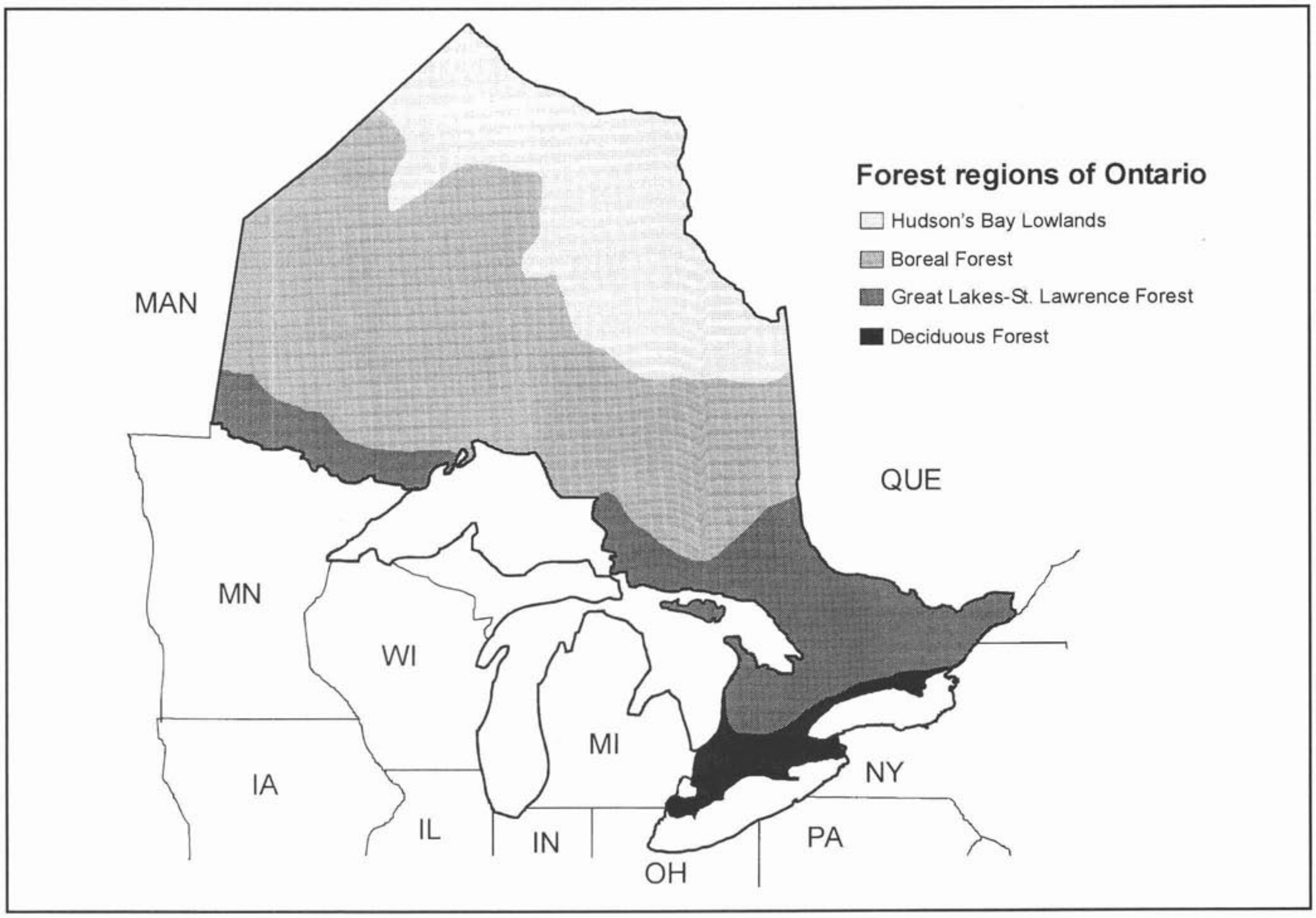

Fig. 1. Forest regions of Ontario (adapted from OMNR (1996)).

(Kimmins and Lavender 1987, Singh and Wheaton 1991, Freedman *et al.* 1992, Hogg and Hurdle 1995, Colombo *et al.* 1998, Thompson *et al.* 1998). In this report, we address the potential effects of elevated atmospheric CO_2 concentrations and climate change on Ontario's forest ecosystems. A general discussion of the potential environmental, social, and economic impacts of climate change in Ontario is presented by Smith *et al.* (1998). We confine our discussion to the response of forest vegetation to climate change and evaluate silvicultural activities that may be used to: (1) reduce the negative impacts of climate change on forests (principally trees) and (2) increase C storage by forests.

Forest Regions of Ontario

Ontario has a total area of 107 million ha, about 74% of which is forested. This 79 million ha represents 17% of Canada's and 1% of the world's forests (OMNR 1996). Ontario's forests are represented by the Hudson's Bay Lowlands and the Boreal, Great Lakes-St. Lawrence (GLSL), and Deciduous Forest Regions (Fig. 1). The Hudson Bay Lowlands contain 24 million ha, roughly half of which is classified as "productive" forest land (i.e., capable of growing commercial tree species), the rest being non-forested due to its arctic climate. The Boreal Forest Region is the largest forested area in Ontario, covering an area of 43 million ha between the Hudson Bay Lowlands and the GLSL forest. This region is dominated by black spruce (*Picea mariana* (Mill.) BSP), white spruce (*Picea glauca* (Moench) Voss), jack pine (*Pinus banksiana* Lamb.), balsam fir (*Abies balsamea* L.), aspen (*Populus* sp.), and white birch (*Betula papyrifera* Marsh.). The GLSL forest occupies 9 million ha laying south of the Boreal Forest Region and extends west of Thunder Bay, north of Lake Huron, through central Ontario to the St. Lawrence River valley. These forests are dominated by eastern white pine (*Pinus strobus* L.), red pine (*Pinus resinosa* Ait.), and several tolerant and mid-tolerant hardwood species (e.g., sugar maple (*Acer saccharum* Marsh.), red maple (*A. rubrum* L.), yellow birch (*Betula alleghaniensis* Britt.), and red oak (*Quercus rubra* L.)). The Deciduous Forest Region originally encompassed 3 million ha in the southwest portion of the province. However, agricultural and urban development have reduced these forests to 15% of the original area, most of which is in Provincial Parks or small isolated stands on private lands. The Deciduous Forest Region contains a variety of hardwood species common to the Eastern Deciduous Forest of the United States (Elliott 1998).

Ontario's Climate in the 21st Century

General Circulation Models (GCMs), sophisticated computer models, are the primary tool used to predict future climate. Numerous GCMs are used (Lau *et al.* 1996), including a Canadian GCM (Boer *et al.* 1992, McFarlane *et al.* 1992). Most GCMs have outputs for present day ($1 \times CO_2$) and

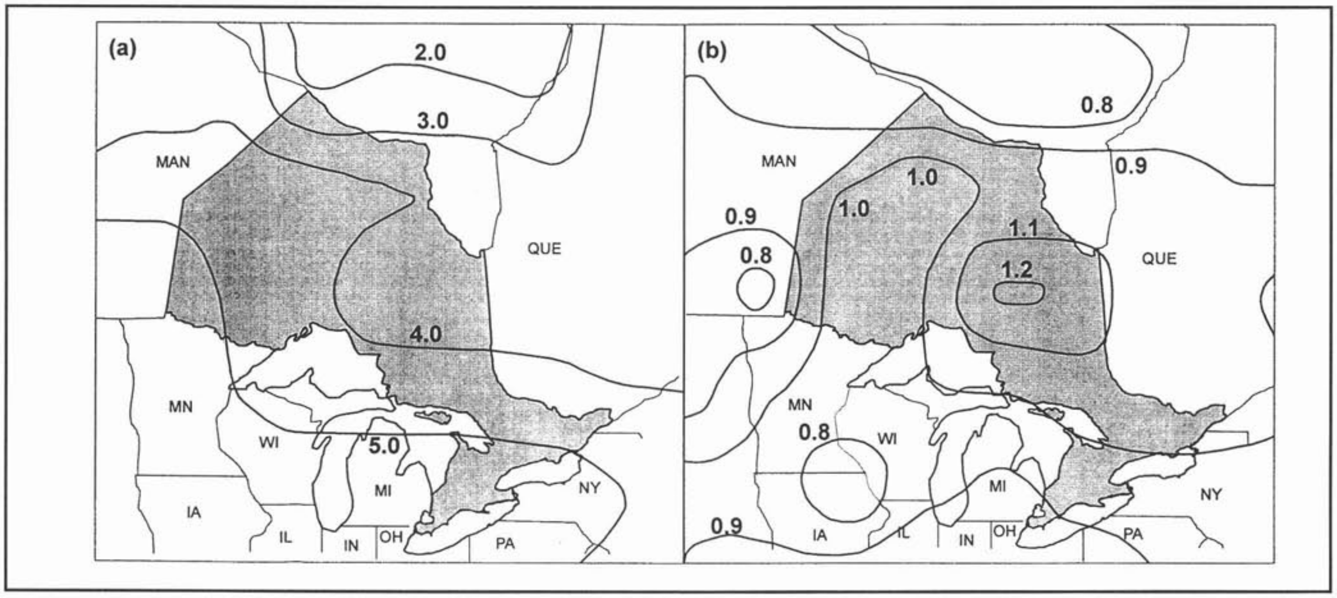

Fig. 2. Predicted (a) mean increases in air temperature (°C) and (b) precipitation ratio during the growing season (1 May – 31 August) for Ontario under a doubled compared to present day atmospheric CO_2 concentration. Black lines in (b) represent the ratio of predicted to current precipitation. Values >1.0 indicate increased precipitation, while values <1.0 indicate reduced precipitation (from Colombo *et al.* (1998)).

doubled ($2 \times CO_2$) CO_2 scenarios, which roughly correspond to atmospheric CO_2 conditions in the mid-20th century and those anticipated for the second half of the 21st century, respectively.

The Canadian GCM predicts that a doubling of CO_2 will increase mean surface air temperature in Ontario by 3–5°C during the growing season (1 May – 31 August) (Fig. 2a) and alter regional precipitation regimes (Fig. 2b). The largest temperature increases are expected in southern Ontario and southwestern sections of northwestern Ontario. The smallest temperature increases of just under 3°C are anticipated over extreme northwestern Ontario near Hudson's Bay. A doubling of CO_2 is projected to decrease growing season precipitation by 10% over much of northwestern and southern Ontario and increase by 10–20% in northeastern Ontario (Fig. 2b). Higher temperatures will increase evapotranspiration and lead to drier soils in areas where precipitation is reduced or unchanged, such as in northwestern and south central Ontario. In northeastern Ontario, where both growing season temperature and precipitation are predicted to increase, only minimal negative climatic effects on site water balance may occur.

Climate Change, Disturbance, and Vegetation Dynamics of Ontario's Forests

Elevated CO_2 and associated changes in climate can influence forest ecosystems directly through effects of temperature, precipitation, and length of the frost-free period on physiological and ecological processes (Bazzaz 1996, Saxe *et al.* 1998). In addition, changes in climate variability can be critical to ecosystem distribution, structure, and function because many of the ecological impacts of climate are the result of extremes. Climate also influences forests indirectly by affecting the frequency and severity of disturbances; the increased frequency of disturbance by fire, insects, and extreme weather expected with climate change will exert a stronger effect on forest ecosystems than the effects of modified climate alone (Davis and Botkin 1985, Overpeck *et al.* 1990, Bazzaz 1996).

Climatic Disturbances

Large regional changes in average temperature and precipitation are anticipated in Ontario under a doubled CO_2 scenario (Fig. 2). More importantly, extreme weather events are expected to be more frequent (Francis and Hengeveld 1998) since higher temperatures will increase energy flow through the climate system and modify the hydrological cycle and other physical processes that drive weather events. A number of significant regional trends in abnormal precipitation patterns (e.g., drought, periods of heavy rainfall, severe winter storms) have appeared since 1970 (Francis and Hengeveld 1998). However, comparison of the current and projected future frequency distributions of mean summer air temperature shows that small changes in average temperature greatly increase the probability of abnormally warm summers, even if the variability in mean temperature remains the same (Fig. 3). In this example, an increase in average summer temperature from 15.3 to 16.9°C increases the likelihood of having an average summer temperature of 17.3°C from 1.3% (once in 75 years) to 33% (once every 3 years).

The increasingly warmer and drier climate of some regions will begin to influence forest ecosystems as existing stands become less adapted to the prevailing conditions. Changes in temperature and precipitation, coupled with a higher probability of hot, dry periods, may result in episodes of regional decline in Ontario's boreal and GLSL forests (Millers *et al.* 1989, LeBlanc and Foster 1992, Reed and Desanker 1992). Forest decline, the reduction in growth and vigour and/or widespread mortality of trees over large geographic regions, is a progressive event that occurs over several years (Millers *et al.* 1989). Forest decline is associated with past management practices, stand and site conditions, and/or uncommon climate events (e.g., severe droughts, late spring frosts) that reduce tree vigour (Millers *et al.* 1989). Secondary attack of weakened trees by insects and diseases is common, and often results in significant mortality (Wargo and Harrington 1991).

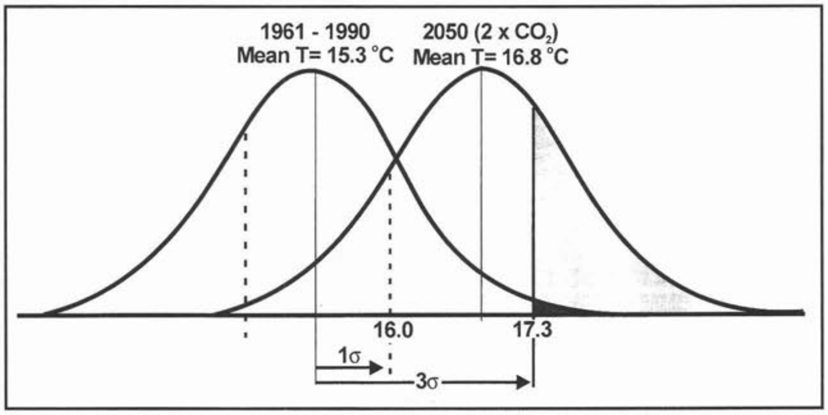

Fig. 3. Frequency distribution (with standard deviations) of mean summer temperature for 1961–1990 and hypothetical doubled CO_2 scenario in 2050. (adapted from Francis and Hengeveld (1998)).

Insects

Insects are an important disturbance agent in Ontario's forests that reduces growth and causes selected mortality of host species or stand-replacing mortality over large areas (Volney 1996). Moderate to severe insect defoliation has affected an average of 15.9 million ha yr^{-1} of forest in Ontario since 1975 (Fig. 4a). Due to the cyclic nature of insect infestations, damage ranged from about 3 to almost 40 million ha yr^{-1} over this period. Primary agents of this damage are spruce budworm (*Choristoneura fumiferana* Clem.), forest tent caterpillar (*Malacosoma disstria* Hubner), jack pine budworm (*Choristoneura pinus pinus* Freeman), and gypsy moth (*Lymantia dispar* L.).

It is generally assumed that climate change will increase insect populations. Because insects are poikilothermic (cold-blooded), warmer but sublethal temperatures will enhance their development, reproduction, distribution, and migration. Temperature and other climatic factors also affect insects indirectly through their influence on host plant physiology, synchrony of plant-insect interactions, and the abundance of their natural enemies (predators, parasites, and diseases) (Wallner 1987, Porter *et al.* 1991, Fleming and Volney 1995). However, because of the complex and poorly understood interactions of insects in forests, only their general responses to climate change can currently be made (Porter *et al.* 1991, Volney 1996).

Insects are highly adaptive and capable of rapid migration to regions or habitats that become favourable as a result of changing climate. If higher winter minimum temperatures in the future reduce overwinter insect mortality, northward migration of non-native species may introduce new insects to Ontario (Kurz *et al.* 1995). Elevated winter minimum temperatures could favor the migration of the mountain pine beetle (*Dendroctonus ponderosae* Hopk.), a serious insect pest of western Canada, into north-western Ontario (Safranyik1990) and cause large losses of pine forests in this region. Migration of insect species may also release them from control by natural enemies, allowing their population size to increase (Porter *et al.* 1991). Since the distribution of insect species is closely tied to the distribution of their host plants, climate change-induced shifts in the regional abundance of a host plant species will likely be reflected in parallel shifts in associated insects (Porter *et al.* 1991).

Climate change may increase the risk of forest disturbance by insects through environmental effects on host tree physiology and health. Spruce budworm, forest tent caterpillar, and jack pine budworm capitalize on environmental conditions that reduce the vigour of the plant host (Wallner 1987, Volney 1996). Outbreaks of these three insect species are likely to increase in regions where climate change results in warmer, drier growing seasons, e.g., northwest Ontario (Fleming and Volney 1995, Volney 1996). Damage by gypsy moth could increase in southern Ontario if the incidence of fall drought and above-average spring temperatures increases (Millers

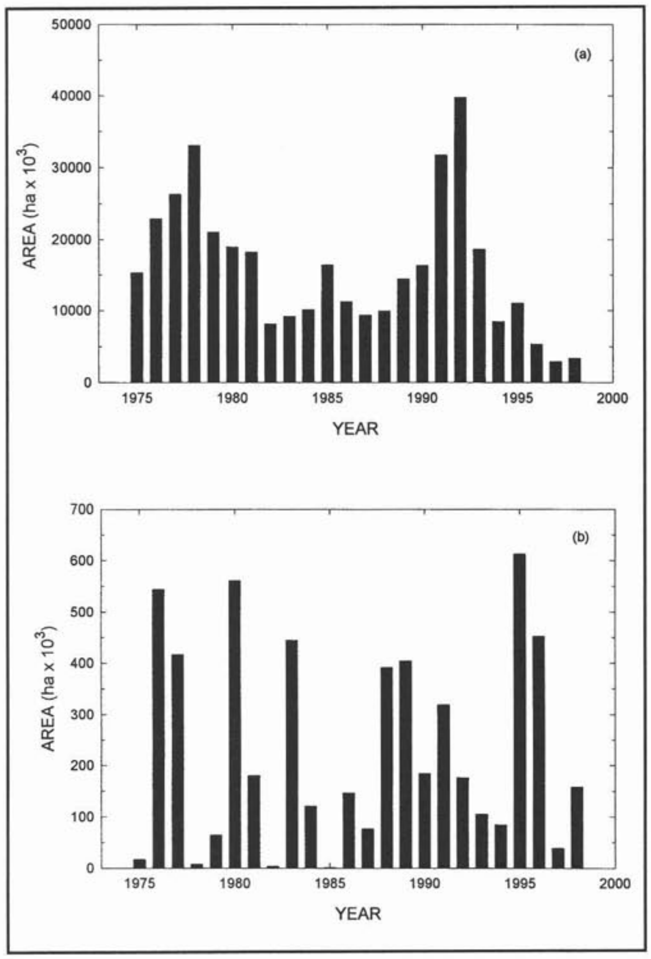

Fig. 4. Land area of Ontario's forests (a) suffering > 30% defoliation of current foliage to insects and (b) burned by wildfire annually from 1975–1998 (CCFM 1999).

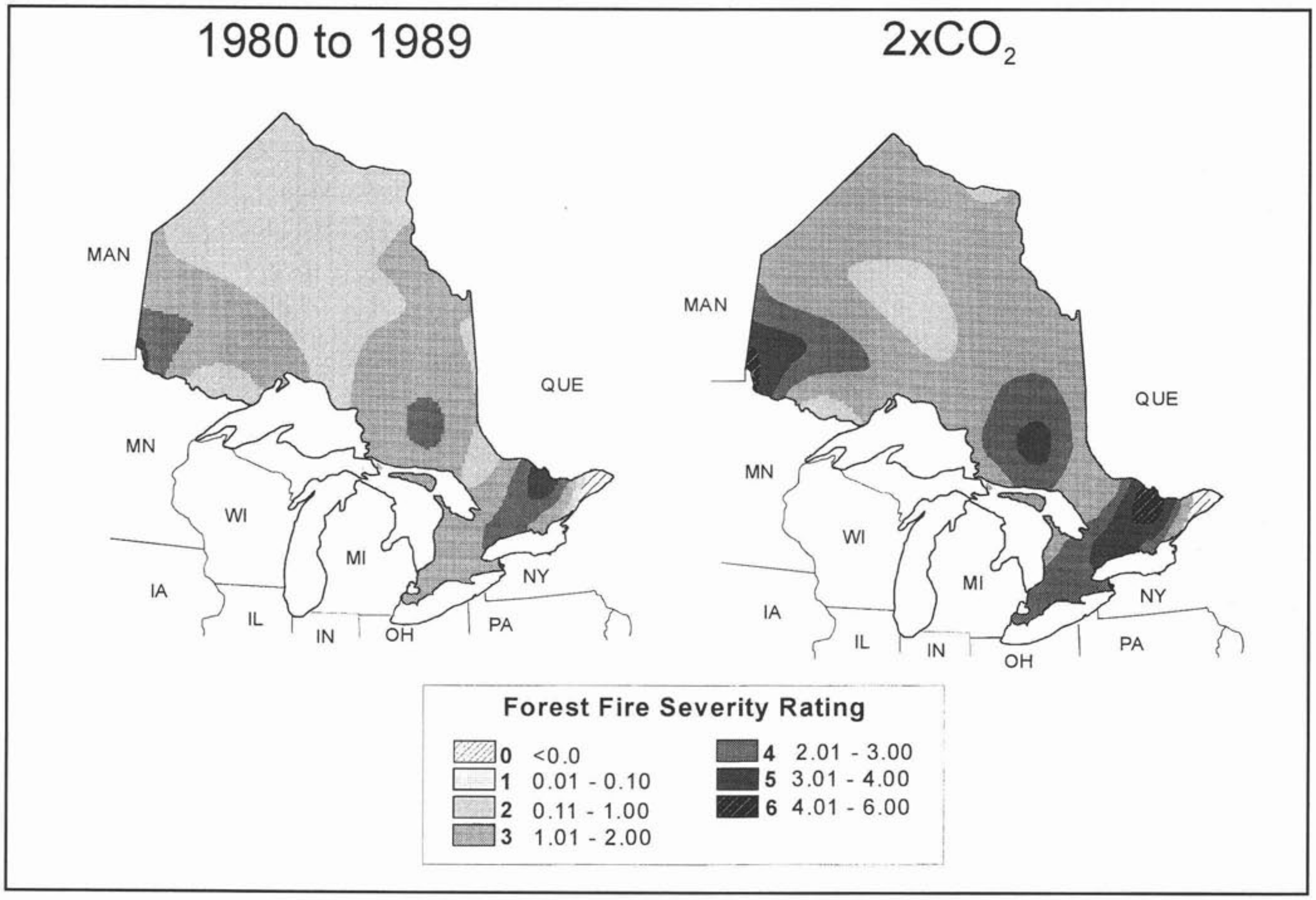

Fig. 5. Seasonal forest fire weather severity rating for Ontario from 1980 to 1989 compared with a doubled CO_2 scenario. Fire weather severity rating is an index calculated by integrating the effects of temperature, precipitation, humidity and wind velocity for a given time period. Larger values correspond to higher fire weather severity (from Colombo *et al.* (1998)).

et al. 1989). Elevated CO_2 alone may alter leaf chemistry and the nutritional quality of leaves, although the effects this may have on insect feeding guilds and insect interactions with natural enemies are unclear (Porter *et al.* 1991, Lindroth *et al.* 1993, Bezemer and Jones 1998).

Fire

Wildfire is the dominant natural disturbance in Ontario's forest ecosystems, burning a total of 5.5 million ha since 1975. The area burned annually is variable, but averaged 197 000 ha yr^{-1} over the past 30 years (Fig. 4b). Increases in evapotranspiration and reductions in precipitation are expected to increase fire weather severity and the area burned annually in Canada by about 40% (Flannigan and Van Wagner 1991). There is a strong consensus that climate change will also increase fire activity in most of Ontario (Simard 1997) due to a higher frequency and severity of drought years (Fig. 5). The most pronounced increase in fire weather severity is expected in the extreme northwest and south-central regions of the province. The fire season in Ontario may also be about 25 days (16%) longer due to elevated spring and fall temperatures (Wotton and Flannigan 1993). The change in climate could further heighten the risk of wildfire because of increased lightning activity (Price and Rind 1994), currently the ignition source for most of the area burned in Ontario. Larger forest floor fuel loads in declining stands or where insect infestations have occurred will predispose these forests to stand-replacing fire, particularly where fire weather severity has increased.

Forest Vegetation

Changes in climate along with a doubling of atmospheric CO_2 are expected to dramatically alter the composition and extent of Ontario's forests (Solomon 1986, Pastor and Post 1988, Sargent 1988, Solomon and Bartlein 1992, Lenihan and Neilson 1995, Mackey and Sims 1993). The largest relative changes in vegetation will be seen at the margins of existing forest regions: the northern tree line of the boreal forest, the forest-prairie ecotone in northwestern Ontario, and at the interface between the southern limit of the boreal forest and the northern limit of the GLSL forest. Higher temperatures and altered precipitation regimes are projected to initiate the northward recession of the southern border of the boreal forest, as species poorly adapted to the new climate become less competitive, decline, and in severe instances, die. The new climate of this region will be better suited to GLSL species and initiate their northern migration and replacement of boreal species.

An increase of up to 3.5°C in temperature over the next 50–100 years will shift the climatic range of species approximately 100–500 km to the north. However, northward migration is unlikely to keep pace with retreat at the southern margins of forest regions. Inadequate natural dispersal rates, lack of suitable soils, increased frequency of disturbance, landscape fragmentation due to agricultural and urban development, and natural barriers to plant migration (e.g., the Great Lakes) are expected to hinder the northward movement of forest species (Davis *et al.* 1986, Roberts 1989, Loehle and LeBlanc 1996, Iverson and Prasad 1998). As a result, forest regions may decrease in size and become fragmented (Solomon and Bartlein 1992, Lenihan and Neilson 1995, Weber and Flannigan 1997). Climate change and CO_2 enrichment may also change species' present competitive relationships and result in new species combinations with no modern counterparts (Loehle and LeBlanc 1996, Bazzaz 1996, Alward *et al.* 1999). As a result, whole-scale replacement of the boreal by the GLSL forest type in its current form is unlikely, a conclusion supported by palaeoecological reconstructions that suggest species migrate singly rather than as intact plant communities (Peters 1990).

The composition of Ontario's naturally regenerated forests 50 or more years from now is difficult to predict. In regions where large changes in climate occur over a short time span, generalist species (those in which a typical genotype is adapted to a wide range of environmental conditions), such as red pine and red maple, will be favoured over more specialized species. Species most susceptible to the effects of climate change are those that are localized, highly specialized, or poorly adapted for long distance dispersal. Isolated populations or those located at the edge of a species range are most at risk (Peters and Lovejoy 1992).

Changes in forest tree species composition will tend to lag behind that of changes in climate due to the longevity of trees and impediments to migration (Davis and Botkin 1985). The anticipated increase in disturbances by fire, insects, and extreme weather, however, will accelerate changes in forest composition and favor species with high migration rates, that reach sexual maturity early, and that are adapted to disturbance. In regions where fire is expected to increase in frequency, fire-adapted tree species will be favoured. If the fire interval is shorter than the age of sexual maturity of tree species, the migration of tree species (natural or human-assisted) will be hindered. This will enhance the establishment of fire-adapted hardwood and herbaceous species, and increase the likelihood of conversion of conifer forests to scattered hardwood forests, shrub woodlands, or grasslands in the area of transition between the boreal and GLSL forests (Long and Hutchin 1991, Bazzaz 1996). In this scenario, jack pine forests in northwestern Ontario may be replaced by grasslands or by aspen parklands currently located in relatively dry, southern portions of the Prairie provinces (Hogg and Hurdle 1995, Schindler 1998). Higher fire frequency in the Ottawa River valley could hinder the current successional replacement of white pine and red oak forests by more fire-sensitive, shade-tolerant hardwood species. Conversely, in regions at the interface of the boreal and GLSL forest types where fire frequency declines, less fire-adapted GLSL species, such as white pine, red pine, eastern white cedar (*Thuja occidentalis* L.), and balsam fir, could increase in abundance (Suffling 1995, Flannigan *et al.* 1998).

Effects of Climate Change on Tree Growth and Forest Productivity in Ontario

Carbon is assimilated by plants through photosynthesis, allocated to various plant parts, and is either incorporated into new tissues, or re-enters the atmosphere through respiration. In ecological terms, forest productivity is defined by net primary production (NPP), the accumulation of C in living plant tissues (i.e., photosynthesis minus respiration) per unit land area over a given time interval. From a forestry perspective, however, forest or stand productivity is defined in terms of the capacity to produce stem wood or fibre for use in forest products. In this context, forest productivity is measured in terms of annual increment in volume or basal area of stem wood per unit land area and accounts for both additions and losses to growing stock.

Climate warming in a CO_2-enriched atmosphere will influence forest productivity through effects on the physiological processes affecting tree growth. Elevated temperatures will lengthen the growing season of temperate and boreal forests due to an earlier onset of warm temperatures in the spring and a delay in frosts and cold temperatures in the fall. Slight to moderate increases in temperature will also enhance tree metabolic processes and growth in a direct manner. However, since woody plant respiration increases exponentially with temperatures between 10°C and 25°C (Kozlowski and Pallardy 1997), large increases in air and soil temperatures may reduce tree growth and forest productivity by increasing respiratory losses of above- and below-ground plant tissues.

Elevated CO_2 will improve the productivity of all Ontario forest plants by increasing the net rate of photosynthesis (P_n) (Table 1). The physiological response to CO_2 is reduced somewhat under resource- limited conditions (e.g., low light, water, or nutrients), but will be generally larger in: (1) deciduous compared to coniferous species, (2) younger than older trees, and (3) earlier successional species (Körner 1993, Curtis 1996, Curtis and Wang 1998, Saxe *et al.* 1998). On average, tree seedlings exposed to doubled CO_2 and plentiful light, water, and nutrients increase 40 to 50% in P_n, decrease 20% in respiration, and increase 20 to 50% in growth rate (Curtis 1996, Curtis and Wang 1998, Saxe *et al.* 1998).

Higher plants are generally more resource-use efficient in a CO_2 enriched atmosphere, with less light, nitrogen (N), and water required per unit C fixed in photosynthesis (Drake *et al.* 1996). Elevated CO_2 results in higher net C fixation, improved light use efficiency and shade tolerance, and increased response of P_n to temperature, such that the CO_2 enhancement of light limited and light saturated rates of P_n will be comparatively larger in warmer climates projected for the future (Drake *et al.* 1996). Elevated CO_2 also results in increased P_n per unit N, a mineral that commonly limits growth of northern forests. Higher P_n combined with a slight reduction (~10%) or no change in leaf conductance to water vapour in a CO_2-enriched atmosphere will improve water use efficiency and perhaps drought tolerance.

Modest (<15%) to substantial (20–50%) (Melillo *et al.* 1993, Cao and Woodward 1998, Pan *et al.* 1998) increases in NPP have been predicted for northern forests under doubled CO_2, largely dependent upon N availability (Melillo *et al.* 1993, McGuire *et al.* 1995). However, growth losses and mortality due to drought and more frequent disturbance by fire, insects, and disease may offset potential gains in NPP and wood production due to elevated CO_2. Although a rigorous prediction of the future productivity of Ontario's forest must await development of regional climate models and provincially based ecosystem models, some general statements regarding the productivity of Ontario's forests in a changed climate can

Table 1. Possible effects of altered environment induced by climate change on stand/forest productivity and timber supply in northwestern northeastern, and south-central Ontario in the next 50 to 100 years[a].

	Stand/forest productivity in regions of Ontario		
Processes and factors	**Northwest**	**Northeast**	**South-central**
Environment:			
Elevated temperature	+	+	+
Increased CO_2	+	+	+
N availability	+	+	0
Soil moisture availability	– –	+	–
Increased herbaceous competition	–	– –	–
Fire disturbance	– –	–	– –
Insect damage	– –	0	–
Disease	– –	0	–
Net effects on stand/forest productivity	–	+	0
Net effects on regional timber supply	– –	+	–

[a]Anticipated effects on forest productivity due to changes in processes and factors in the right column are designated as positive (+), none (0), negative (–), and strongly negative (– –).

be made (Table 1). On all but the poorest sites, productivity will tend to increase as a result of higher temperature and atmospheric CO_2 concentration. Warmer spring and fall temperatures will lengthen the growing season, while elevated CO_2 and higher temperatures will increase P_n and resource-use efficiency. Nitrogen availability in soils will tend to increase with temperature as a result of higher rates of mineralization and decomposition, although the effect could be relatively less in the south due to the warmer temperatures already prevailing there. However, reduced precipitation and increasing severity of drought in the northwest and south could counter the positive effects of warmer temperature, elevated CO_2, and N availability. In contrast, if precipitation increases in the northeast as predicted, there could be large increases in stand productivity. More frequent disturbance will also reduce productivity in some regions, depending on the relative changes in the frequency and intensity of these events.

Management Approaches to Adapt Ontario's Forests to Climate Change

Silvicultural Systems and Stand Management

Maintaining forest vigour during progressive disequilibrium between forest vegetation and climate will require silvicultural systems that address the management and regeneration of declining stands. In stands not ready for commercial harvest, thinning or selective removal of suppressed, damaged, or poor quality individuals can be used to increase light, water, and nutrient availability and the vigor of the residual overstory (Smith *et al.* 1997). Thinning effects on insect populations will need to be considered so as not to create environmental and stand conditions that favour insect pests. This approach will also require efforts to minimize logging damage and windthrow in the residual stand and to protect desirable advance reproduction. Where the understory species are unacceptable as a source of regeneration, underplanting with site-adapted species, species mixtures, or genotypes may be preferred. In older declining stands, logging prior to stand deterioration followed by planting can be used to speed the establishment of better-adapted forest types. Partial cutting to improve stand vigour is likely a viable option for treatment of declining stands of GLSL forest types, as it is consistent with the ecology and management of mid-tolerant and shade-tolerant GLSL species. In comparison, partial cutting and underplanting of forest types such as black spruce may not be ecologically or economically viable.

Regeneration and Stand Establishment

Regenerating Ontario's forests in a changing climate will present many new challenges to forest managers. Most important among these could be extensive artificial regeneration efforts to assist the migration of tree species and genotypes from their present to future ranges (Davis 1989, Mackey and Sims 1993). The timing of such initiatives should be determined based on the performance of selected species and genotypes in experimental plantings and provenance trials on sites north of their current range. The northward movement of certain species will in some instances be hindered by the lack of suitable soil types and the expected lag in soil development during climate change, such as where the climate will allow more nutrient-demanding hardwoods to be planted on more northerly, acidic, less fertile sites now occupied by conifers. Planting hardwoods on nutrient-poor sites may be facilitated by mixed planting of hardwoods with N-fixing species (e.g., alder (*Alnus* sp.)), planting hardwood species inoculated with site-adapted mycorrhizae, or forest fertilization.

Vegetation management treatments will need to be tailored to control plant species that become more competitive in a climate change environment. Early successional, herbaceous weed species with high sexual and vegetative reproductive potential (e.g., *Calamogrostis* sp., bracken fern (*Pteridium aquilinum* (L.) Kuhn), *Bromus* sp., and raspberry (*Rubus* sp.)) can be expected to increase in abundance in Ontario with climate change because they are adapted to disturbance, exhibit a large response to CO_2 enrichment compared to trees, and their short life cycles favour rapid genetic adaptation to new climatic conditions (Körner 1993, Bazzaz 1996). Broadleaved woody species adapted to a wide variety of site environments (e.g., red maple) and/or to disturbance (e.g., aspen and white birch) will likely become stronger competitors for conifers. Changes in interspecific competition among tree species under elevated CO_2 could result in the development of novel forest types, requiring new silvicultural approaches for their management (Davis 1989, Bazzaz 1996). The ongoing appearance of new mixes of forest tree and other plant species may continue for centuries before more stable, steady-state forest environments occur and migration (natural and assisted) has brought plant species into equilibrium with site conditions.

Climate-induced effects on the abundance and diversity of soil microbial communities may influence the ability of ecosystems to adapt to changing climatic conditions (O'Neill 1994, Wullschleger *et al.* 1994). Silvicultural practices that help to maintain the diversity of soil micro-organisms, such as mycorrhizal fungi, can be used to improve regeneration efforts. Practices such as clearcutting, whole-tree harvesting, and prescribed burning can decrease the diversity and total biomass of mycorrhizae, important to water and nutrient uptake of tree seedlings in drier environments (Harvey *et al.* 1980). Reducing the size of clearcuts and maximizing cut block edges can be used to promote the ingress of mycorrhizae from surrounding undisturbed stands (Harvey *et al.* 1980). Mixed plantings of conifers and hardwoods can also increase microbial diversity (Hendrickson *et al.* 1982).

Genetic Management

A forward-looking genetic management program for forest species is needed to respond to climate change. Species will differ in the amount of intervention needed to ensure they are adapted genetically to new forest conditions. Widely distributed species (e.g., black and white spruce) will require less attention because of the relatively high level of diversity already present, although some local ecotypes may be threatened and genetic diversity reduced unless steps are taken to identify and conserve them. Maintaining species of economic importance but with currently diminished abundance (e.g., white pine, red pine, and red oak) may require assessment of their genetic diversity to determine the limits of transferability of these genotypes. Climate change will also require periodic updating of climate-based seed zones.

Given the current uncertainty regarding regional shifts in climate, the planting of nursery stock representing widely adapted populations and diverse seed source mixtures has been recommended to increase the likelihood of regeneration success and long-term site adaptation (Ledig and Kitzmiller 1992). Breeding programs promoting genetic diversity, pest resistance, tolerance of environmental stresses, and increased growth with elevated CO_2 may be needed to ensure that species are adapted to the future environment (Namkoong 1984, Wang *et al.* 1995).

Climate change is a serious threat to some rare plant species, often characterized by specialized environmental requirements and low genetic diversity (Peters 1990). Red mulberry (*Morus rubra* L.) and red spruce (*Picea rubens* Sarg.), comparatively rare tree species in Ontario, are sometimes protected by establishing small, isolated natural areas or ecological reserves, centred around limited, disjunct populations. This by itself may be insufficient to prevent extinction of such populations, particularly if climate change causes more frequent disturbances and reduces the environmental suitability of these preserved habitats (Burton *et al.* 1992). Establishment of a larger number of reserves to maintain a variety of habitat conditions, interconnected by migration corridors, may reduce the risk of local extinction (Peters 1990). However, because of the small population size of such species and the slow migration rates of trees, such corridors may prove unsuccessful. Protection of woodland herbs will be particularly difficult because of their low reproductive output and poor adaptation for long distance seed dispersal (Cain *et al.* 1998). Attempts to save locally rare species and genotypes should, however, be viewed in the continental context of abundance or rarity, instead of using provincial abundance as a cause for initiating costly conservation programs.

If a decision is made to protect a rare or threatened species, intensive, costly silvicultural activities may be needed if suitable habitat becomes difficult to find or create due to climate change. In light of such decisions, conservation plans and innovative silvicultural approaches will be needed to sustain these species, and should include breeding plans that incorporate systems to increase genetic diversity (Namkoong 1984). However, our ability to only forestall rather than prevent local extinction needs to be weighed against the cost of preservation activities. In addition, severe habitat loss could mean that long-term preservation of some rare species may only be possible in "zoo-like" arboreta, rather than in natural environments.

Biodiversity, "the variability among living organisms and the ecological complexes of which they are a part," can be viewed in the context of three elements: "ecosystems, species, and genes" (CCFM 1997). The conservation of biodiversity is an important goal of sustainable forest management, but this goal may be threatened by climate change (Peters 1990, Hebda 1996). Should climate change alter an aspect of the habitat crucial to the survival of a species, biodiversity will change regardless of forest management activities. One way to maintain the current level of biodiversity (at least temporarily) is to protect old-growth forests and other unique ecosystems as reservoirs of biodiversity (Burton *et al.* 1992). However, the ability to retain these ecosystems is likely to decline as the climate changes and the frequency and severity of forest disturbance increases.

Forest Protection

Increased future forest fire activity in some regions of Ontario will require expanded fire protection and fire risk mitigation efforts (Stocks *et al.* 1998, Wilson and Baker 1998) to prevent an increase in the area burned. The predicted increase in wildfire may overwhelm the current resources of fire management agencies, as is already observed in years with high fire activity, and would jeopardize future timber supply and place unique ecosystems at risk. Even with the practice of "modified protection," where the ecological role of fire is allowed to run its course, greater efforts to protect high-value areas will be needed. Investments required to prevent the area burned from increasing need to be balanced against the level of protection desired to protect specific resources, given their economic or social value (Stocks *et al.* 1998).

Forests can be protected from decline-related insect and disease outbreaks by using silvicultural practices that maximize stand or tree growth, remove at-risk stands or trees, alter insect/disease habitat, or increase diversity (Smith *et al.* 1997). "Stress management" practices, such as partial cutting or thinning, can increase stand vigour and lower the susceptibility to attack (Wargo and Harrington, 1991, Gottschalk 1995). Sanitation cuts can reduce disease losses by removing infected, unhealthy individuals. Since these practices can increase vulnerability to other insects and diseases, site- and species-specific approaches will be needed (Smith *et al.* 1997). Shortening the rotation length can also decrease the period of stand vulnerability to damaging insects and diseases, such as spruce budworm and oak decline (Gottschalk 1995). Other silvicultural techniques that reduce tree stress and susceptibility to insect and disease include regulating species composition (i.e., planting species best adapted to site environment) and maintaining biological diversity. Where silvicultural activities for insect pest management are ineffective or inappropriate, and the value of the resource warrants it, the use of insecticides may be considered.

The Forest Carbon Cycle

The global C content is fixed; it occurs as atmospheric CO_2 and oceanic C or in storage as living and dead biomass, wood products, fossil fuels, limestone, and marble. In simple terms, atmospheric CO_2 concentration depends on the balance between the amount of C in storage and C released and trans-

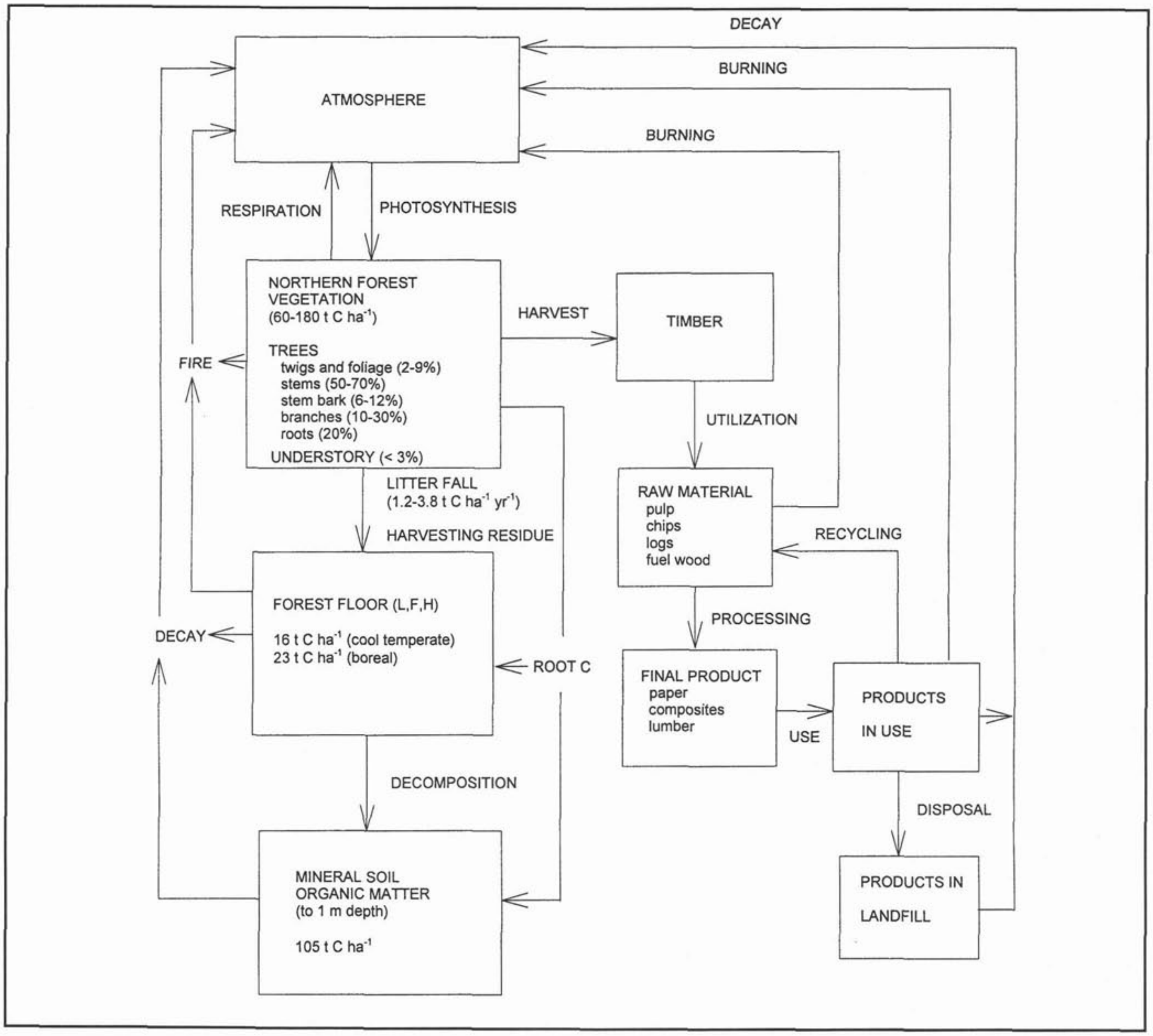

Fig. 6. Generalized forest-wood products cycle for northern forest ecosystems (adapted from Karjalainen *et al.* 1998). Values for forest biomass and soil C are from Schlesinger (1977), Freedman *et al.* (1986), Vogt *et al.* (1986), Morrison (1990), and Grigal and Ohmann (1992).

ferred from storage to the atmosphere by natural processes, disturbances, and human activities (Houghton *et al.* 1983, Amthor 1995). Current global CO_2 emissions are ~7.0 Gt C yr^{-1} (1 gigatonne (Gt) = 1 billion tonnes; 1 tonne (t) = 1000 kg), due primarily to fossil fuel burning, deforestation, and land use changes. About 46% of these CO_2 emissions remain in the atmosphere, resulting in a present annual increase in atmospheric CO_2 concentration of 1–2 ppmv (Keeling *et al.* 1995).

Forests occupy roughly 4.1 billion ha (26%) of the earth's land surface and play a prominent role in the global C cycle (Dixon *et al.* 1994). About 77% of the total terrestrial C is in forests (1150 Gt C), roughly two-thirds of which (800 Gt C) occurs in forest floor detritus and mineral soil organic matter (Dixon *et al.* 1994, Amthor 1995). Forests account for 90% of the annual C flux between terrestrial ecosystems and the atmosphere through photosynthesis (80–120 Gt C absorbed yr^{-1}), respiration (35–55 Gt C released yr^{-1}), and decomposition of litter and soil organic matter (~45 Gt C released yr^{-1}) (Amthor 1995). The large size of forest C pools and fluxes means that even small changes in the global forest C cycle can significantly (positively or negatively) influence atmospheric CO_2 concentration.

Forest C is located in above- and below-ground living and dead biomass, forest floor detritus, mineral soil, and, after harvest, in wood products (Fig. 6). Total forest ecosystem C and the size of forest C pools vary with climate and site factors influencing photosynthesis, C allocation patterns in plants, and C losses through respiration, decomposition, herbivory, and combustion by fire. Cool temperate and boreal forests typically contain 60–180 t C ha^{-1} in above- and below-ground biomass, and an additional 116–149 t C ha^{-1} in the forest floor and mineral soil (Schlesinger 1977, Post *et al.* 1982, Prentice and Fung

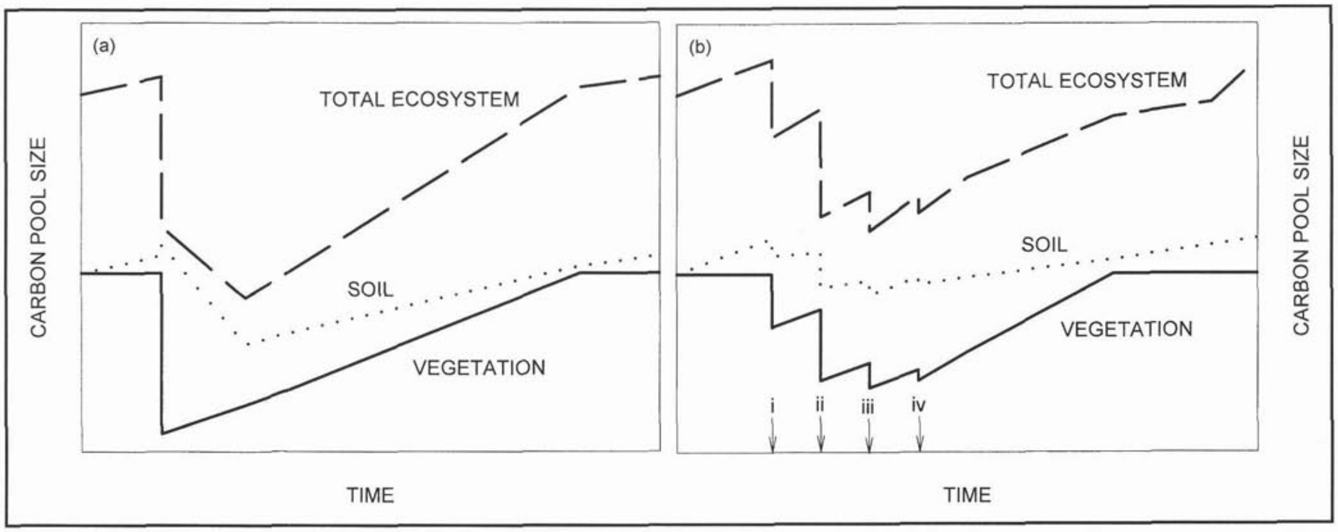

Fig. 7. Relative changes in forest vegetation (solid line), soil (dotted line), and total ecosystem (dashed line) C pools with time following (a) clearcut or (b) uniform shelterwood harvest. A 100 year rotation was assumed for the clearcut. Shelterwood harvest assumed to include preparatory (i), seed (ii), regeneration release (iii), and complete removal (v) harvests at 20 year intervals beginning at stand age of 80 yr. Arrows indicate time of the four shelterwood harvests. Scarification to create seedbed for natural regeneration was assumed to coincide with the seed cut.

1990). Northern forested peatlands have deep (2–4 m) organic soils containing large amounts of C, and average 120 t C ha^{-1} in forest biomass and 1300 t C ha^{-1} in peat (Zoltai and Martikainen 1996).[5]

The size of forest ecosystem C pools also varies with time after disturbance and the intensity of disturbance (Aber *et al.* 1978, Harmon *et al.* 1990, Pastor and Post 1986). Biomass accumulation and growth of even-aged forests increase with leaf area development, reach a maximum rate at peak stand leaf area, and commonly undergo an age-related decline (Ryan *et al.* 1997). Following a stand-replacing disturbance (e.g., wildfire, harvest), biomass is greatly reduced and soil C losses increased due to higher decomposition rates (Fig. 7a). Since the biomass and NPP of young, regenerating forests are initially low, total ecosystem C declines after disturbance until biomass accumulation and litter inputs to the soil exceed soil C losses to decomposition. In northern hardwood forests, 20–50% of the forest floor litter mass is decomposed within 10–30 years of clearcut harvesting, followed by a gradual increase of soil C to its pre-harvest level within 50–60 years (Covington 1981) (Fig. 7a). Disturbances affecting only a portion of the overstory (e.g., forest decline, shelterwood harvest) have less effect on the size and decay rates of forest C pools due to reduced biomass removal and a lower decomposition rate under a partial canopy (Fig. 7b). Although forest ecosystems exhibit a decline in productivity (e.g., NPP, mean annual stemwood volume increment) with age, total ecosystem C often continues to increase, largely because of C accumulation in forest floor biomass and mineral soil (Fig. 7) (Harmon *et al.* 1990, Morrison 1990).

[5]Ontario contains 22.5 million ha of peatlands (wetlands with peat depth ≥40 cm depth), 7.9 million ha of which are productive black spruce forests (Ketcheson and Jeglum 1972, Zoltai 1988). The effects of climate change and forest management on C dynamics of northern peatlands are addressed by Gorham (1991, 1995).

Harvesting diverts woody biomass into the wood products C pool (Fig. 6). Utilization of biomass has become more efficient over the past century, but 30 to 40% of pre-harvest above-ground biomass of second-growth forests may be left on site as slash (Harmon *et al.* 1996, Winjum *et al.* 1998). The harvested biomass is processed into products having short (e.g., fuel wood, newsprint, packaging) or long (e.g., lumber, plywood, composite materials) decay rates. By-products of manufacturing may be used for other products, or burned for energy, releasing this C to the atmosphere. Wood products decay while in use and are eventually burned, recycled, or placed in landfills, where they decay slowly. Of the 9 Mt C (1 megatonne (Mt) = 1 million tonnes) in wood products produced in Canada in 1990, about 75% had a lifetime longer than 5 years (Winjum *et al.* 1998). Although the wood products C pool is <1% of Canada's total forest C, modifying the fate and decay rate of these products can capture large amounts of C.

Forest Management Options to Increase C Sequestration in Ontario's Forests

As a party to the United Nations Framework Convention on Climate Change (UNFCCC), Canada accepted a greenhouse gas emissions reduction target of 6% below the 1990 level of 567 Mt (CO_2 equivalents) by 2008–2012. From 1990 to 1995, greenhouse gas emissions in Canada increased by 9% to 619 Mt (Jacques *et al.* 1997). At the current rate of increase, total emissions will reach 777 Mt by 2010, or 244 Mt (46%) above the emissions reduction target of 533 Mt. However, Environment Canada (1997) projects 2010 national emissions levels to be 669 Mt, or 136 Mt (26%) above the emissions target, due to a variety of mitigative efforts. Ontario, with its large population (11.4 million, ~38% of Canada's population) and industrial base, contributed ~31% of Canada's 1990 emissions (Jacques *et al.* 1997). Ontario's emissions are predicted to increase by 12% over 1990 levels by 2010 due to continued reliance on fossil fuels for electricity generation (Environment Canada, 1997). National-provincial agreements to reduce greenhouse gas emissions may hold Ontario responsible for their contribution to the national total.

A number of forest management activities hold potential to increase C storage in biomass and mitigate net greenhouse gas emissions. Currently, the Kyoto Protocol recognizes only reforestation and afforestation as accountable activities in efforts to meet emissions reduction targets. However, recognition of other forestry activities that promote C sequestration (e.g., modified harvesting, site preparation, planting, tree improvement, fertilization, and tending) and the role of wood product C pools in this accounting are currently being debated.

Five general mitigation strategies have been identified for forestry and the wood-using industries: (1) creation of new forests, (2) maintenance of existing forests, (3) expansion of existing forest C sinks and pools through modified forest management activities, (4) increased long-term storage of forest C in wood products, and (5) substitution of wood-based fuels for fossil fuels (direct substitution) and substitution of wood for energy-intensive structural materials such as steel, aluminum, concrete, and plastics (indirect substitution) (Brown *et al.* 1996, Kohlmaier *et al.* 1998). These C mitigation strategies aim to increase C storage in forest biomass and wood products and maximize the use of wood in substitution.

In the short-to-medium term, the creation, conservation, and more intensive management of forests have the highest potential to increase C sequestration, particularly when trees harvested are used in durable wood products with slow decay rates. In the longer term, increased substitution is a more sustainable mitigation option, since it emphasizes the transfer of biomass C into products that reduce the amount of fossil fuel burned (Koch 1992, Matthews 1996, Schlamadinger and Marland 1996). Traditionally, larger, higher quality logs were used for long-lived structural materials. However, technological advances in the manufacture of engineered lumber using low quality wood will further increase the amount of biomass that can be used in indirect substitution. Indirect substitution of wood can dramatically reduce CO_2 emissions relative to the use of steel, plastics, and aluminum that require, respectively, 10, 27, and 76 times more energy for their manufacture (Schopfhauser 1998). Wood and biofuels are CO_2-neutral energy sources since their combustion results in no net gain in atmospheric CO_2: the CO_2 released from burning these fuels is cycled back into biomass through photosynthesis (Zerbe 1993). In addition, use of biomass for energy reduces the use of fossil fuels. For example, for each tonne of wood substituted for coal, ~0.47 t of C emissions are avoided (Klass 1998).

A quantitative assessment of C mitigation options must include a complete accounting (i.e., life cycle assessment) of **net** CO_2 emissions associated with all activities within the forest-wood products cycle (see Richter 1998). Site preparation, herbicide and fertilizer application, and forest product manufacturing may increase C sequestration but in carrying out these activities some C is also released to the atmosphere. The net C balance of these options must therefore be weighed against their economic costs and the costs of other non-forest options to reduce atmospheric CO_2 (Rubin *et al.* 1992, van Kooten *et al.* 1992, 1999). The economic costs of forest sector C mitigation options are addressed elsewhere (van Kooten *et al.* 1992, 1999, Hoen and Solbrig, 1994, Dixon 1997, Sedjo *et al.* 1997).

Creation of New Forests

Afforestation to convert low-C storage agricultural lands or old fields to high-C storage forest plantations is an effective and ecologically viable means of increasing C sequestration in the short-to-medium term (Rollinger *et al.* 1997, Winjum and Schroeder 1997, Richter *et al.* 1999). Short rotation forestry using fast growing hardwoods and intensive management can increase C storage several times over that of traditional plantations of native species, particularly where the biomass produced is used for energy. Regardless of the plant material and silvicultural intensity, these new forests require significant investments in establishment, management, and protection from fire, insects and disease, and the development of commercial markets for the biomass produced (e.g., lumber, composite materials, and biofuels) (Samson *et al.* 1999, van Kooten *et al.* 1999).

There are limited opportunities for afforestation in Ontario because of the lack of unforested land and, what unforested land is present is under pressure from agriculture and urbanization (Table 2). Since the climate and soil of the boreal forest region are largely unsuitable for agriculture, most of the land in northwestern and northeastern Ontario is already forested (Table 2). Most marginal agricultural land that might be considered for afforestation is located in southern Ontario. Much of this land was originally forested and is better suited to trees than agricultural crops. However, since most of this land is privately owned, strong markets for wood biomass and other financial incentives will be needed to encourage its afforestation (van Kooten *et al.* 1999).

The contribution of afforestation in Ontario to offset greenhouse gas emissions depends on the potential forest productivity of the available land, the growth rate of the species planted, and the intensity of forest management. Depending on these variables, planting GLSL forest species on 500 000 ha in southern Ontario could offset 1.8–15.4% of the provincial contribution to Canada's emissions reduction target in 2010, while short rotation plantations and intensive culture of willow (*Salix* sp.) or hybrid poplar could offset 3.0–23.1% of these emissions (Table 3). If hybrid poplar biomass was substituted for coal, 0.6–1.7 Mt C emissions could be avoided, greatly increasing the value of afforestation in C mitigation.

Although afforestation can make a significant contribution to provincial efforts to reduce emissions, the obstacles to realizing this potential by 2010 are substantial (Samson *et al.* 1999, van Kooten *et al.* 1999). If the goal was to afforest half of the available lower class agricultural land in Ontario, about 450 000 ha would need to be planted. Starting such a program in 2002 would mean that in the nine years up to and including 2010, an average of 50 000 ha would have to be planted annually. A total of 100 million seedlings or cuttings (in the case of vegetatively propagated willow or poplar) would be required each year, assuming 2000 trees ha^{-1}. The logistical challenges to obtain adequate supplies of (largely) southern seed, site preparation equipment, and planting crews will be substantial. In addition this effort will be complicated by the fact that most land for afforestation is in small, privately owned parcels.

Forest Conservation

The area of North American forests and C they contain has increased over the last century. This is the result of increased emphasis on regeneration, improved silviculture, fire suppression,

Table 2. Land area in developed agriculture, grass and meadow, and productive forest by administrative region. Ownership divided between crown land and other (private and federal). Table does not include non-productive forest land, unclassified or unsurveyed land.

	Land area (1000 hectares)					
	Northeast and northwest regions			South-central region		
Land Classification	Crown	Other	Total	Crown	Other	Total
Developed agriculture	3.8	258.1	261.9 (0.74)[a]	15.3	4356.9	4372.2 (41.6)
Grass and meadow	5.4	70.5	75.9 (0.21)	11.7	901.3	913.0 (8.7)
Productive forest[b]	32 280.2	2803.0	35 083.2 (99.1)	1859.3	3360.0	5219.3 (49.7)
Total			35 421.0			10 504.5

[a]Percent of total land area in a given land classification by region is enclosed in parentheses (OMNR 1996).
[b]Productive forest is land area capable of growing commercial trees.

and succession of abandoned agricultural land to forests (Turner *et al.* 1995, Wernick *et al.* 1998). The area annually harvested in Ontario over the past 25 years has not varied appreciably (~212 000 ha yr^{-1}), with 80% of harvesting occurring in the boreal forest (Fig. 8a). Artificial regeneration has increased during this period, with greater emphasis on planting and less on seeding and natural regeneration (Fig. 8b). Despite improved regeneration efforts, large areas of forest land in Ontario are currently poorly stocked for a variety of reasons (CCFM 1999). Selective reforestation of areas with very low C density (<2 t C ha^{-1}) is a significant, economically viable C mitigation option (Kurz and Apps 1995).

More intensive management of forests and improved efficiency in harvesting, processing, and utilization of forest biomass have also reduced the forest area needed to produce a given unit of wood products. For example, improved forest management and better wood utilization in the United States have allowed forest area to remain unchanged and standing timber volume to increase by about 30% since 1900, despite a 70% increase in the volume of timber harvested annually (Wernick *et al.* 1998). These practices also allow conservation of forests with high C densities, such as old growth or peatland forests (Brown *et al.* 1996, Zoltai and Martikainen 1996), or management of more forests for non-timber assets (e.g., wildlife, biodiversity, and recreation).

The higher frequency of disturbance by fire, insects, and extreme weather with climate change seriously threatens Ontario's boreal forests and will require heightened protection efforts if future wood supply and C stored in these forests is to be maintained or increased above current levels. Increases in fire activity will require greater fire suppression efforts so that as much forest C as possible can be harvested rather than lost to the atmosphere by combustion. Forest management techniques to maintain the health of existing forests and to decrease C emissions associated with climate change-induced forest decline are also needed. Without such protective measures, fire and insect disturbance could act as a positive feedback to climate change and accelerate CO_2 release and warming of the atmosphere (Kurz *et al.* 1995, Schindler 1998).

Table 3. Annual above-ground C storage and contribution of afforestation of 500 thousand ha in southern Ontario to greenhouse gas emission reduction targets under two future emission scenarios.

Species/forest type	Mean annual above-ground production[a] (t C ha^{-1})	Percent contribution[b]	Percent contribution[c]
White pine	0.8–3.1	1.8–7.4	2.9–11.9
Red pine	1.1–2.9	2.6–7.0	4.2–11.3
Red oak	1.3–4.0	3.0–9.5	4.8–15.4
Intolerant hardwoods	1.0–3.0	2.4–7.1	3.8–11.5
Tolerant hardwoods	0.8–3.7	1.9–8.8	3.1–14.2
Willow	3.5–6.0	8.3–14.3	13.5–23.1
Hybrid poplar	1.3–3.5	3.0–8.3	4.8–13.5
Hybrid poplar w/ substitution for coal[d]	1.3–3.5	5.8–16.2	9.4–26.2

[a]Mean annual increment in aboveground production (stems, branches, foliage) of GLSL forest species/types from Alban *et al.* (1978), Freedman *et al.* (1986), Alban and Perala (1992), Gower *et al.* (1993), and Reich *et al.* (1997). Ranges in values reflect differences in site quality and management intensity. Mean annual increment in aboveground production (stems and branches) for short rotation willow (12 000–15 000 stems ha^{-1}; 3–4 year harvest cycle) and hybrid poplar (1100–1400 stems ha^{-1}; 10–15 year harvest cycle) populations estimated from experience in Ontario and Quebec (Samson *et al.* 1999).
[b]"Business as usual" scenario: Canada's greenhouse gas emissions are 212 Mt C in 2010 (68 Mt C above the Kyoto target) and Ontario contributes 31% (21 Mt C) of this 68 Mt C emissions gap. Emissions in 2010 estimated using linear regression and data from 1990–1995 (Jacques *et al.* 1997).
[c]Environment Canada (1997) scenario: Canada's greenhouse gas emissions are 182 Mt C in 2010 (37 Mt C above the Kyoto target) and Ontario contributes 36% (13 Mt C) of this 37 Mt C emissions gap.
[d]Wood produced in hybrid poplar plantations used in substitution for coal: 1 t of poplar wood is energy equivalent of 0.67 t bituminous coal, and 1 t of coal releases 0.71 t C when burned (Klass 1998, van Kooten *et al.* 1999).

Forest Management to Increase C Storage

More intensive management of Ontario's forests can increase C storage in biomass and soils through activities that promote C uptake by trees. In most cases, forest management can increase C storage by using existing silvicultural techniques on a larger scale. For example, site preparation, assisted natural regeneration, planting, and vegetation management can increase the rate of C accumulation and shorten the rotation needed to obtain a given volume of wood. Forest management practices such as fertilization (on some soils) and tree improvement can also potentially enhance C storage by increasing stand and site productivity (Farnum *et al.* 1983). Further, forest management to increase C storage need not be practised to the detriment of other forest objectives. Ecologically sustainable forestry practices, such as reduced impact logging, can be used to increase C storage as well as to reduce environmental damage, protect biodiversity, and retain wildlife habitat. These activities may in some cases serve the dual purpose of increasing forest health and adapting species composition to a warmer, drier climate, as discussed earlier.

Harvesting

Forest management alters the natural C cycling between forest ecosystems and the atmosphere, in many cases reducing maximum C storage and increasing C emissions to the atmosphere relative to unmanaged forests. Because forests accumulate C beyond the age of maximum sustained yield (i.e., maximum mean annual increment) or maximum financial return, managed forests generally contain one-third to one-half less C than unmanaged forests (Cooper 1983, Harmon *et al.* 1990). As a consequence, lengthening the rotation increases C storage. However, the comparative potential and associated costs of longer rotations to increase forest C storage depend on the natural disturbance interval and life-span of the forest type in question (Cooper 1983, Harmon *et al.* 1990, Price *et al.* 1997). Conversion of long-lived (>200 years old) coastal Douglas-fir-western hemlock (*Pseudotsuga menziesii* (Mirb.) Franco – *Tsuga heterophylla* (Raf.) Sarg.) forest ecosystems to intensively managed, commercial forests on a rotation (70–80 year) much shorter than the natural disturbance interval, significantly reduces their total C storage (Harmon *et al.* 1990). Conversely, in boreal forest regions where the natural disturbance interval is less than the age of maximum sustained yield, additional C storage can be obtained by extending rotation length only at the cost of protection from fire (Price *et al.* 1997). Where the costs of protection are prohibitive, shorter rotations to sequester C in wood products is obviously preferable to the loss of stored forest C to the atmosphere by fire (Price *et al.* 1997).

The value of simply extending rotations to increase the amount of C stored in forests must be balanced against ecological, technical, economic, and social considerations. Where lengthening the rotation is not a viable option, the net C sequestered can be increased if the C released from decaying wood products from the previous rotation is reduced relative to the C accumulated in the forest over the next rotation (Price *et al.* 1997, Winjum *et al.* 1998). This improvement in C storage can be accomplished by planting fast-growing species or genotypes, intensive forest management, and/or using wood in longer-lived forest products. Where tree biomass is used for energy production, very short rotations (3–15 years) can be a more effective mitigation option than traditional forest management for wood production (Dewar 1990, Alban and Perala 1992, Schlamadinger and Marland 1996).

From an operational perspective, the optimal rotation age is dependent on forest type, site productivity, and the desired forest product. From 1990–1997, ~55% of wood harvested in Ontario was used for lumber and veneer, 39% for pulp and paper and 9% for fuelwood (CCFM 1999). Clearcutting and even-aged management are used for jack pine, upland black spruce, and aspen forests. Management of these species on relatively short rotations may be preferable to extending rotations to increase ecosystem C, particularly where there is a significant risk of stand loss due to fire (Price *et al.* 1997). Longer rotations may be a more practical means of increasing C storage in black spruce lowland forests because the current and future risk of fire on these sites is lower. Carbon storage in longer-lived GLSL conifer and tolerant hardwood forest types managed with partial cutting systems can be enhanced by increasing the age (and therefore the size) at which residual overstory trees are removed, or by permanent retention of a portion of the large, old canopy trees.

Silvicultural Systems

Harvesting intensity (i.e., the proportion of stand basal area harvested) can influence the net C balance of managed forests over a rotation (Fig. 7) (Cooper 1983). Assuming similar end uses for wood, a clearcut silvicultural system has the largest C removal per unit land area and rotation due to complete overstory removal and the initial reduction in forest floor biomass due to lower litter inputs and increased soil organic matter decomposition. Partial cutting systems (e.g., shelterwoods and selection harvesting) reduce the net C output relative to clearcutting, since significant amounts of biomass are always present on a site as large trees or regeneration (Fig. 7); disturbance and exposure of the forest floor are also reduced, and where advance reproduction is present, site preparation and planting may not be required. However, these gains are partially offset by the higher frequency of return to harvest (i.e., C emissions associated with harvesting), lower efficiency of harvest (i.e., fewer m^3 of wood ha^{-1}), and higher risk of logging damage to residual trees and advance reproduction. On the positive side, large amounts forest floor C may accumulate at higher residual overstory densities due to lower rates of decomposition and higher litter inputs, particularly in long-lived tolerant hardwood forests managed using single-tree selection (Piene and Van Cleve 1978, Rollinger *et al.* 1997). However, harvesting intensity apparently has little long-term impact on total ecosystem C provided the forest is regenerated quickly allowing litter inputs to the soil to recover (Aber *et al.* 178, Harmon *et al.* 1990, Dewar and Cannell 1992).

Regeneration and Early Stand Treatments

Forestry practices that reduce the time from harvesting to crown closure (i.e., the regeneration phase) increase the rate of C uptake per hectare and total C accumulation per rotation (Houghton *et al.* 1983, Kurz and Apps 1995, Smith *et al.* 1997). Prompt regeneration through planting or use of advance reproduction, rather than reliance on newly established natural regeneration, shortens the period of early stand establishment when C accumulation is low and soil C losses relatively high (Schroeder 1991, Hoen and Solbrig 1994). Use of artificial rather than natural regeneration may reduce the regeneration phase by 5–10 years (Kurz and Apps 1995). Planting of high quality nursery stock to control initial density and spacing also reduces the regeneration phase since fuller, more uniform stocking and crown closure are achieved earlier than with newly established natural regeneration (Smith *et al.* 1997). In overly dense, naturally regenerated stands, density regulation before competition-induced mortality and growth stagnation of the self-thinning stage occurs can increase the rate and amount of C stored per rotation (Schroeder 1991). Tree improvement programs that provide stress resistant, fast-growing, or CO_2-responsive genotypes for planting, can further reduce the length of the regeneration phase and increase total C storage.

Control of competing vegetation by site preparation and release treatments can improve C storage by increasing seedling survival, shortening the regeneration phase, and increasing light, water, and nutrient availability (Walstad and Kuch 1987, Schroeder 1991, Dixon 1997). Vegetation management treatments to improve stand establishment can increase stem biomass two- to six-fold within 7–15 years of treatment, and therefore represent a significant forest management option for

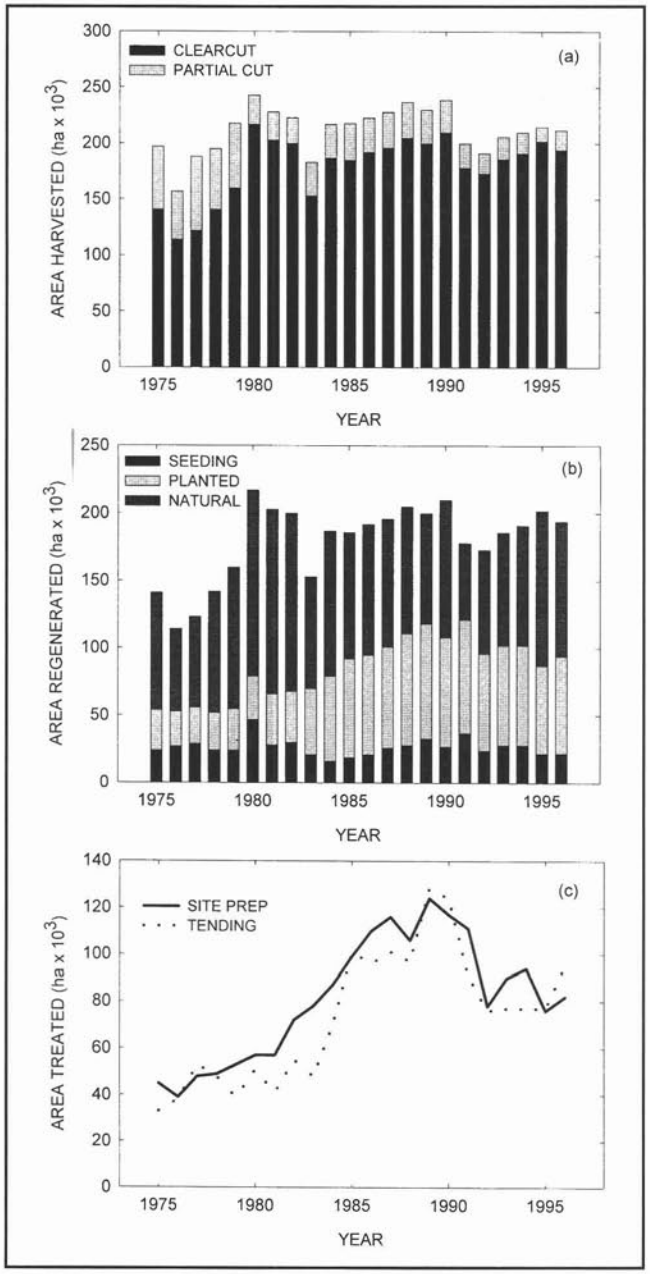

Fig. 8. Silvicultural statistics for total land area (a) harvested, (b) regenerated, and (c) site prepared or tended in Ontario from 1975 to 1996. Partial cutting refers to lands harvested using seed tree, shelterwood, and selection systems. Site preparation refers to use of mechanical, prescribed fire, chemical, or hand treatments that modify the site to provide favourable conditions for natural and artificial regeneration. Tending refers to any activity carried out to benefit a crop at any stage of its life, and includes thinning, improvement cuts, and release from competing vegetation (does not include site preparation) (CCFM 1999).

increasing the rate of C storage (Stewart *et al.* 1984, Newton *et al.* 1987). From 1989–1996, nearly 89 000 ha were site prepared annually in Ontario, 44% of the annual area harvested over this period (Fig. 8). Mechanical scarification accounted for 82% of site preparation, with lesser areas treated with herbicide (13%) and prescribed burning (5%) (CCFM 1999). Release treatments were annually applied to an area equal to 40% of that harvested from 1989–1996. Herbicide was the most popular means of release, being used on 95% of the area treated in Ontario (CCFM 1999). Use of these early stand treatments on a greater proportion of area harvested could have large positive impacts on C storage.

Despite disturbance of the forest floor, the long term impact of scarification and prescribed burning on forest C balance is minimal unless these treatments are unnecessarily severe (Jurgensen *et al.* 1990, Johnson 1992). Mechanical site preparation methods that incorporate organic matter or logging residues into the soil (e.g., disking and mulching) or disturb only small patches (e.g., cultivation using the Bräcke) help retain soil C and improve the growth of regeneration and are preferred to methods that remove or displace this material (e.g., scalping large areas and windrowing). Use of herbicides in site preparation and release is less disruptive of the forest floor and likely has little effect on the release of forest soil C.

Thinning

Thinning of forest stands can increase the vigour, growth, and final merchantable volume and biomass of individual residual trees, but total stand stem wood volume or biomass are commonly unchanged or reduced (Cooper 1983, Dewar and Cannell 1992). This is because thinning simply concentrates growth on fewer stems without increasing the inherent productivity of the site. Thinned stands also have lower forest floor and soil C relative to unthinned stands because of decreased amounts of woody litter (Dewar and Cannell 1992). As a result, thinned stands typically contain less C in stem biomass and forest floor regardless of the fate of harvested material (Schroeder 1991, Dewar and Cannell 1992). Thinning is rarely used in Ontario but could help to increase net C sequestration when an increase in volume and quality of residual trees leads to production of durable wood products or their use in indirect substitution.

Fertilization

Fertilization can increase tree growth and total C storage of forest plantations by increasing the inherent productivity of a site and enhancing the photosynthetic response to elevated CO_2 (Huettl and Zoettl 1992, Dixon 1997). Growth responses to fertilization are generally greatest on lower quality sites, but vary with species, stand age, and stand density (Foster and Morrison 1983, Schroeder 1991, Bell *et al.* 1997). Although thinning alone does not increase net C storage, fertilization of thinned stands can enhance stand growth by reducing tree mortality from competition and allowing for crown expansion compared with fertilization of fully stocked, unthinned stands (Farnum *et al.* 1983, Schroeder 1991). Fertilization and planting N-fixing species also often increase soil C, as well as improving forest growth and aboveground biomass production (Johnson 1992).

The productivity of northern coniferous forest ecosystems is commonly limited by N, although positive growth responses to K and P fertilization have been reported (Foster and Morrison 1983). However, forest fertilization is not presently used in Ontario and was practised only on a very small scale in Ontario in the past (< 1% of forest land was fertilized from 1983 to 1992) (Bell *et al.* 1997). Past research on eastern Canada's tree species has focussed on mid-rotation N fertilization of semi-mature jack pine, with average five-year volume gains of 6–10 $m^3 ha^{-1}$ (i.e., 1.2–2.0 t C ha^{-1}) reported (Foster and Morrison 1983, Morrison and Foster 1990, Bell *et al.* 1997). Positive five-year volume responses of black spruce (2.2 $m^3 ha^{-1}$), white spruce (4.5 $m^3 ha^{-1}$), red spruce (4.6 $m^3 ha^{-1}$), and balsam fir (7.1 $m^3 ha^{-1}$) have been exhibited, but fertilization responses of these species have been inconsistent (Foster and Morrison 1983). Further research on the fertilization response to site, age, and stand density for Ontario's forest tree species is needed to evaluate the use of fertilization in mitigation (Morrison and Foster 1990, Bell *et al.* 1997).

To represent a viable mitigation option the gain in forest ecosystem C storage from fertilization must exceed the C emitted in the production, transportation, and application of fertilizer. Emissions of N_2O in ammonia production, and NO_x and N_2O emissions from fertilizer volatilization in the field, all of which contribute to the greenhouse effect, must also be considered when determining net C balances (Hoen and Solbrig 1994). The chemical reactions involved in the manufacture of one unit of ammonia, the primary component of nitrogen fertilizer, produces about four units of CO_2 (Sittig 1979). This emission is offset by a volume gain of 1–2 $m^3 ha^{-1}$ per fertilizer application[6] (Schroeder 1991, Hoen and Solbrig 1994), suggesting that use of ammonia fertilizers will likely be an effective C mitigation method where the growth response is large enough to justify the investment.

Conclusions

Human impacts on the global C cycle are expected to double the atmospheric CO_2 concentration by the middle of the 21st century and to profoundly affect the earth's climate (Houghton *et al.* 1983, Vitousek 1994, IPCC 1996). In response to the projected effects of climate change, the 1992 UNFCCC agreed to stabilize greenhouse gas concentrations in the atmosphere. In 1997, as a party to the UNFCCC, Canada accepted a greenhouse gas emissions target of 6% below its 1990 level, to be met by 2008-2012. Canada is examining the role of forestry to achieve its international obligation to reduce greenhouse gas emissions.

The formulation and timely application of forest management responses to a changing climate will be needed to minimize the adverse impacts of climate change on Ontario's forests. This will require the development of regional climate models and models to predict the effects of elevated atmospheric CO_2 and various climate change scenarios on forest C budgets as well as the distribution, structure, and function of future Ontario forest ecosystems. Understanding and recognizing the potential effects of climate change on forests will allow resource managers to begin to modify forest management planning and policies to lessen negative impacts on the sustainability, biodiversity, health, and socioeconomic benefits of forest ecosystems. Resource managers will also need to view forests as idiosyncratic and prone to new and unexpected behaviour to ensure that existing models, policies, and programs are resilient to anticipated impacts of climate change.

Finally, a better understanding of forest ecosystem function will help us formulate forestry's role in C mitigation. At present, more intensive forest management during stand establishment represents the single best method of increasing C storage in Ontario's forests. Forest management practices such as planting, density regulation, and vegetation management can significantly increase the rate and amount of forest C storage. Emphasis on these forestry practices to increase C uptake is also consistent with Ontario's forest legislation, policy, standards, and guidelines directed at the sustainability of forest ecosystems. As such, more intensive forest management will provide a number of additional environmental, economic, and social benefits to the people of Ontario. Use of forest management for C mitigation also provides a measure of flexibility given the uncertainty regarding the rate and impact of climate change: short-term reliance on Ontario's forest sector can also help to avoid the economic hardship of meeting emission targets largely through reduced fossil fuel consumption, buying time to develop and adopt more energy-efficient technologies and carbon-free energy sources.

Carbon sequestration through afforestation has significant potential to contribute to 2010 emissions reduction targets but this option is constrained logistically. However, the establishment of new forests for C sequestration in southern Ontario has additional short-term, non-carbon benefits, such as improving biodiversity, providing wildlife habitat, protecting watersheds, increasing recreational opportunities, and contributing to the restoration efforts in the Deciduous Forest Region (Freedman and Keith 1996). In the longer term, continued development of the infrastructure and technology needed for cost effective, short rotation biomass energy plantations could reduce Ontario's reliance on fossil fuels.

Acknowledgements

We thank the following colleagues for their contributions to this report: Don Higgs, Celia Graham, Jane Nicholson, Dr. Michael Ter-Mikaelian, Trudy Vaittinen, and Lisa Buse of the Ontario Ministry of Natural Resources, and Mike Wotton, Canadian Forest Service. Special thanks to Dr. David Layzell of BIOCAP Canada for discussions and information regarding the Kyoto Protocol, trends in Canada's greenhouse gas emissions, and afforestation. We also thank two anonymous reviewers for their valuable comments on an earlier version of this manuscript.

References

Aber, J.D., D.B. Botkin and J.M. Melillo. 1978. Predicting the effects of different harvesting regimes on forest floor dynamics in northern hardwoods. Can. J. For. Res. 8: 306–315.

Alban, D.H. and D.A. Perala. 1992. Carbon storage in Lake States aspen ecosystems. Can. J. For. Res. 22: 1107–1110.

Alban, D.H., D.A. Perala and B.E. Schlaegel. 1978. Biomass and nutrient distribution in aspen, pine, and spruce stands on the same soil type in Minnesota. Can. J. For. Res. 8: 290–299.

Alward, R.D., J.K. Detling and D.G. Michunas. 1999. Grassland vegetation changes and nocturnal global warming. Science 283: 229–231.

Amthor, J.S. 1995. Terrestrial higher-plant response to increasing atmospheric [CO_2] in relation to the global carbon cycle. Global Clim. Change 1: 243–274.

Bazzaz, F.A. 1996. Plants in Changing Environments. Cambridge University Press, Cambridge, UK.

Bell, F.W., P.J. Bastarache and L. Meyer. 1997. Jack pine fertilization: a review of the literature. Ont. Min. Natur. Resour. Northwest Sci. & Technol. Thunder Bay, Ont. TR-111. 18 p.

Bezemer, T.M. and T.H. Jones 1998. Plant-insect herbivore inter-

[6]Assuming a typical 224 kg N ha^{-1} application, average wood specific gravity for conifers of 400 kg m^{-3}, and stem biomass is 50% C by dry mass.

actions in elevated CO_2: quantitative analyses and guild effects. Oikos 82: 212–222.

Boer, G.J., N.A. McFarlane and M. Lazare. 1992. Greenhouse gas-induced climate change simulated with the CCC second-generation General Circulation Model. J. Clim. 5: 1045–1077.

Brown, S., J. Sathaye, M. Cannell and P.E. Kauppi. 1996. Mitigation of carbon emissions to the atmosphere by forest management. Common. For. Rev. 75: 80–91.

Burton, P.J., A.C. Balisky, L.P. Coward, S.G. Cumming and D.D. Kneeshaw. 1992. The value of managing for biodiversity. For. Chron. 68: 225–237.

Cao, M. and F.I. Woodward. 1998. Dynamic responses of terrestrial ecosystem carbon cycling to global climate change. Nature 393: 249–252.

Canadian Council of Forest Ministers (CCFM). 1997. Criteria and indicators of sustainable forest management in Canada. Technical Report 1997. Natur. Resour. Can., Can. For. Serv., Ottawa, ON. 137 p.

Canadian Council of Forest Ministers (CCFM). 1999. National Forestry Database Program, (http://www.nrcan.gc.ca.cfs/proj/iepb/nfdp/cp95/data_e/tab61e_2.htm).

Cain, M.L., H. Damman and A. Muir. 1998. Seed dispersal and the holocene migration of woodland herbs. Ecol. Monogr. 68: 325–247.

Colombo, S.J., M.L. Cherry, C. Graham, S. Greiffenhagen, R.S. McAlpine, C.S. Papadopol, W.C. Parker, T. Scarr, M.T. Ter-Mikaelian and M.D. Flannigan. 1998. The impacts of climate change on Ontario's forests. Ontario Min. Natur. Resour., For. Res. Inf. Rep. No. 143. 50 p.

Cooper, C.F. 1983. Carbon storage in managed forests. Can. J. For. Res. 13: 155–166.

Covington, W.W. 1981. Changes in forest floor organic matter and nutrient content following clear cutting in northern hardwoods. Ecology 62: 41–48.

Curtis, P.S. 1996. A meta-analysis of leaf gas exchange and nitrogen in trees grown under elevated carbon dioxide. Plant Cell Environ. 19: 127–137.

Curtis, P.S. and X. Wang. 1998. A meta-analysis of elevated CO_2 effects on woody plant biomass, form and physiology. Oecologia 113: 299–313.

Davis, M.B. 1989. Lags in vegetation response to greenhouse warming. Clim. Change 15:75–82.

Davis, M.B. and D.B. Botkin. 1985. Sensitivity of cool-temperate forests and their fossil pollen record. Quat. Res. 23: 327–340.

Davis, M.B., K.D. Woods, S.L. Webb and R.P. Futyma. 1986. Dispersal versus climate: Expansion of *Fagus* and *Tsuga* into the Upper Great Lakes region. Vegetatio 67: 93–103.

Dewar, R.C. 1990. A model of carbon storage in forests and forest products. Tree Physiol. 6: 417–428.

Dewar, R.C. and M.G.R. Cannell. 1992. Carbon sequestration in the trees, products and soils of forest plantations: an analysis using UK examples. Tree Physiol. 11: 49–71.

Dixon, R.K. 1997. Silvicultural options to conserve and sequester carbon in forest systems: preliminary economic assessment. Crit. Rev. Environ. Sci. Tech. 27: S139–S149.

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

Drake, B.G., M.A. Gonzalez-Meler and S.P. Long. 1996. More resource efficient plants: A consequence of rising atmospheric CO_2? Annu. Rev. Plant Physiol. Plant Molec. Biol. 48: 609–639.

Elliott, K.A. 1998. The forests of southern Ontario. For. Chron. 74: 850–854.

Environment Canada. 1997. Canada's Second national Report on Climate Change. "Actions to meet commitments under the United Nations Framework Convention on Climate Change," Environ. Can., Min. Environ., Ottawa, Canada. 141 p.

Farnum, P., R. Timmis and J.L. Kulp. 1983. Biotechnology of forest yield. Science 219: 694–702.

Flannigan, M.D. and C.E. Van Wagner. 1991. Climate change and wildfire in Canada. Can. J. For. Res. 21: 66–72.

Flannigan, M.D., Y. Bergeron, O. Englemark and B.M. Wotton. 1998. Future wildfire in circumboreal forests in relation to global warming. J. Veg. Sci. 9: 469–476.

Fleming, R.A. and W.J.A. Volney. 1995. Effects of climate change on insect defoliator population processes in Canada's boreal forest: some plausible scenarios. Water, Air, Soil Poll. 82: 445–454.

Foster, N.W. and I.K. Morrison. 1983. Soil fertility, fertilization and growth of Canadian forests. Dept. Environ., Can. For. Serv., Inf. Rep. O-X-353. 21 p.

Francis, D. and H. Hengeveld. 1998. Extreme weather and climate change. Environ. Can., Min. Environ., Ottawa, Canada. 31 p.

Freedman, B. and T. Keith. 1996. Planting trees for carbon credits: a discussion of context, issues, feasibility, and environmental benefits. Environ. Rev. 4: 100–111.

Freedman, B., P.N. Duinker and R. Morash. 1986. Biomass and nutrients in Nova Scotian forests, and implications of intensive harvesting for future site productivity. For. Ecol Manage. 15: 103–127.

Freedman, B., F. Meth and C. Hickman. 1992. Temperate forest as a carbon-storage reservoir for carbon dioxide emitted by coal-fired generating systems. A case study for New Brunswick, Canada. For. Ecol. Manage. 55: 15–29.

Gorham, E. 1991. Northern peatlands: role in the carbon cycle and probable responses to climatic warming. Ecol. Appl. 1: 182–195.

Gorham, E. 1995. The biogeochemistry of northern peatlands and its possible responses to global warming. *In* G.M. Woodwell and F.T. Mackenzie (eds.). Biotic Feedbacks in the Global Climatic System. Will the Warming Feed the Warming? pp. 169–187. Oxford University Press, New York, NY.

Gottschalk, K.W. 1995. Using silviculture to improve health in northeastern conifer and eastern hardwood forests. *In* L.G. Eskew (comp.). Forest Health Through Silviculture. pp. 219–226. USDA For. Serv., Gen. Tech. Rep. RM-267.

Gower, S.T., P.B. Reich and Y. Son. 1993. Canopy dynamics and aboveground production of five tree species with different leaf longevities. Tree Physiol. 12: 327–345.

Grigal, D.F. and L.F. Ohmann. 1992. Carbon storage in upland forests of the Lake States. Soil Sci. Soc. Am. J. 56: 935–943.

Harmon, M.E., W.K. Ferrell and J.F. Franklin. 1990. Effects on carbon storage of conversion of old-growth forests to young forests. Science 247: 699–702.

Harmon, M.E., S.L. Garman and W.K. Ferrell. 1996. Modeling historical patterns of tree utilization in the Pacific Northwest: carbon sequestration implications. Ecol. Appl. 6: 641–652.

Harvey, A.E., M.F. Jurgensen and M.J. Larsen. 1980. Clearcut harvesting and ectomycorrhizae: survival of activity on residual roots and influence on a bordering forest stand in western Montana. Can. J. For. Res. 10: 300–303.

Hebda, R. 1996. Atmospheric change, forests and biodiversity. Environ. Monitor. Assess. 49: 195–212.

Hendrickson, O., J.B. Robinson and L. Chatarpaul. 1982. The microbiology of forest soils: a literature review. Can. For. Serv., Dept. Environ., Petawawa Nat. For. Inst., Inf. Rep. PI-X-19. 75 p.

Hoen, H.F. and B. Solbrig. 1994. Potential economic efficiency of carbon sequestration in forest biomass through silvicultural management. For. Sci. 40: 429–451.

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

Houghton, R.A., J.E. Hobbie, J.M. Melillo, B. Moore, B.J. Peterson, G.R Shaver and G.M. Woodwell. 1983. Changes in the carbon content of terrestrial biota and soils between 1860 and 1980: a net release of CO_2 to the atmosphere. Ecol. Monogr. 53: 235–262.

Houghton, J.T., L.G. Meiro-Filho, B.A. Callander, N. Harris, A. Kattenburg and K. Maskell. 1995. Climate Change 1995 – the science of climate change. Second Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge Univ. Press Cambridge, UK.

Huetll, R.F. and H.W. Zoettl. 1992. Forest fertilization: its potential to increase the CO_2 storage capacity and to alleviate the decline of the global forests. Water, Air, Soil Poll. 64: 229–249.

Intergovernmental Panel on Climate Change (IPCC). 1996. Climate Change 1995 Impacts, Adaptations and Mitigation of Climate Change: Scientific-Technical Analyses. Cambridge Univ. Press, Cambridge, UK.

Iverson, L.R. and A.M. Prasad. 1998. Predicting abundance of 80 tree species following climate change in the eastern United States. Ecol. Monogr. 68: 465–485.

Jacques, A., F. Neitzert and P. Boileau. 1997. Trends in Canada's greenhouse gas emissions (1990–1995). Environ. Can., Min. Environ., Ottawa, Canada. 73 p.

Johnson, D.W. 1992. Effects of forest management on soil carbon storage. Water, Air, Soil Poll. 64: 83–120.

Jurgensen, M.F., A.E. Harvey, R.T. Graham, M.J. Larsen, J.R. Tonn and D.S. Page-Dumroese. 1990. Soil organic matter, timber harvesting and forest productivity. *In* S.P. Gessel, D.S. Lacate, G.F. Weetman and R.F. Powers. (eds.). Sustained Productivity of Forest Soils. pp. 392–415. Proc. 7th N.A. For. Soils Conf., Univ. BC, Fac. of For. Publ., Vancouver, BC.

Karjalainen, T., A. Pussinen, S. Kellomaki and R. Makipaa. 1998. The history and future dynamics of carbon sequestration in Finland's forest sector. *In* G.H. Kohlmaier, M. Weber and R.A. Houghton (eds.). Carbon Dioxide Mitigation in Forestry and Wood Industry. pp. 25–42. Springer-Verlag, Berlin.

Keeling, C.D., T.P. Whorf, M. Whalen and J. van der Plicht. 1995. Interannual extremes in the rate of rise of atmospheric carbon dioxide since 1980. Nature 375: 666–670.

Ketcheson, D.E. and J.K. Jeglum. 1972. Estimates of black spruce and peatland areas in Ontario. Can. For. Serv., Dept. Environ., Inf. Rep. O-X-172. 29 p.

Kimmins, J.P. and D.P. Lavender. 1987. Implications of climate change for the distribution of biogeoclimatic zones in British Columbia and for the growth of temperate forest species. *In* D.P Lavender (comp.). Woody Plant Growth in a Changing Chemical and Physical Environment. pp 297–309. Proc. IUFRO Workshop, S2.01-11, Univ. of BC, Vancouver, BC.

Klass, D.L. 1998. Biomass for Renewable Energy, Fuels, and Chemicals. Academic Press, San Diego, CA.

Koch, P. 1992. Wood versus nonwood materials in U.S. residential construction: some-energy-related global implications. For. Prod. J. 42(5): 31–42.

Kohlmaier, G.H., M. Weber and R.A. Houghton. 1998. Carbon Dioxide Mitigation in Forestry and Wood Industry, Springer-Verlag, Berlin.

Körner, C. 1993. CO_2 fertilization: The great uncertainty in future vegetation development. *In* A.M. Solomon and H.H. Shugart (eds.). Vegetation Dynamics and Global Change. pp. 53–70. Chapman and Hall, New York, NY.

Kozlowski, T.T. and S.G. Pallardy. 1997. Physiology of Woody Plants, second edition. Academic Press, San Diego, CA.

Kreileman, G.J.J. and A.F. Bouwman. 1994. Computing land use emissions of greenhouse gases. Water, Air, Soil Poll. 76: 231–258.

Kurz, W.A. and M.J. Apps. 1995. An analysis of future carbon budgets of Canadian boreal forests. Water, Air, Soil Poll. 82:321–331.

Kurz, W.A., M.J. Apps, B.J. Stocks and W.J.A. Volney. 1995. Global climate change: disturbance regimes and biospheric feedbacks of temperate and boreal forests. *In* G.M. Woodwell and F.T. Mackenzie. (eds.). Biotic Feedbacks in the Global Climatic System. Will the Warming Feed the Warming? pp. 119–133. Oxford University Press, New York, NY.

Lau, K.-M., J.-H. Kim and Y. Sud. 1996. Intercomparison of hydrologic processes in AMIP GCMs. Bull. Am. Meterol. Soc. 77: 2209–2227.

LeBlanc, D.C. and J.R. Foster. 1992. Predicting effects of global warming on growth and mortality of upland oak species in the midwestern United States: A physiologically based dendroecological approach. Can. J. For. Res. 22: 1739–1752.

Ledig, F.T. and J.H. Kitzmiller. 1992. Genetic strategies for reforestation in the face of global climate change. For. Ecol. Manage. 50: 153–169.

Lenihan, J.M. and R.P. Neilson. 1995. Canadian vegetation sensitivity to projected climate change at three organizational levels. Clim. Change 30: 27–56.

Lindroth, R.L., K.K. Kinney and C.L. Platz. 1993. Responses of deciduous trees to elevated atmospheric CO2: productivity, phytochemistry, and insect performance. Ecology 74: 763–777.

Loehle, C. and D. LeBlanc. 1996. Model-based assessments of climate change effects on forests: a critical review. Ecol. Model. 90: 1–31.

Long, S.P. and P.R. Hutchin. 1991. Primary production in grasslands and coniferous forests with climate change: an Overview. Ecol. Appl. 1: 139–156.

Mackey, B.G. and R.A. Sims. 1993. A climatic warming analysis of selected boreal tree species, and potential responses to global climate change. World Res. Rev. 5: 469–487.

Matthews, R. 1996. The influence of carbon budget methodology on assessments of the impacts of forest management on the carbon balance. *In* M.J. Apps and D.T. Price. (eds.). Forest Ecosystems, Forest Management and the Global Carbon Cycle. pp. 233-243. NATO ASI Series, Vol. I 40, Springer-Verlag, Berlin.

McFarlane, N.A., G.J. Boer, J.-P. Blanchet and M. Lazare. 1992. The Canadian Climate Centre second-generation General Circulation Model and its equilibrium climate. J. Clim. 5:1013–1044.

McGuire, A.D., J.M. Melillo and L.A. Joyce. 1995. The role of nitrogen in the response of forest net primary production to elevated atmospheric carbon dioxide. Annu. Rev. Ecol. Syst. 26: 473–503.

Melillo, J.M., A.D. McGuire, D.W. Kicklighter, B. Moore III, C.J. Vorosmarty and A.L. Schloss. 1993. Global climate change and terrestrial net primary production. Nature 363: 234–240.

Millers, I., D.S. Shriner and D. Rizzo. 1989. History of hardwood decline in the eastern United States. USDA, For. Serv., Gen. Tech. Rep. NE-126. 75 p.

Mooney, H.A., P.M. Vitousek and P.A. Matson. 1987. Exchange of materials between terrestrial ecosystems and the atmosphere. Science 238: 926–932.

Morrison, I.K. 1990. Organic matter and mineral distribution in an old-growth *Acer saccharum* forest near the northern limit of its range. Can. J. For. Res. 20: 1332–1342.

Morrison, I.K. and N.W. Foster. 1990. On fertilizing semimature jack pine stands in the boreal forest of central Canada. *In* S.P. Gessel, D.S. Lacate, G.F. Weetman and R.F. Powers (eds.). Sustained Productivity of Forest Soils. pp. 416–431. Proc. 7th N.A. For. Soils Conf., Univ. B.C., Fac. of For. Publ., Vancouver, BC.

Namkoong, G. 1984. Strategies for gene conservation. *In* C.W. Yeatman, D. Kafton and G. Wilkes (eds.). Plant Gene Resources: A Conservation Imperative. pp. 79–89. AAAS Selected Symposium 87, Boulder, CO.

Newton, M., M.L. McCormack, Jr., R.L. Sidak and J.D. Walstad. 1987. Forest vegetation problems in the northeast and Lake States/Provinces. *In* J.D. Walstad and P.J. Kuch (eds.). Forest Vegetation Management for Conifer Production. pp. 77–103. John Wiley and Sons, New York, NY.

O'Neill, E.G.1994. Responses of soil biota to elevated atmospheric carbon dioxide. Plant Soil 165: 55–65.

Ontario Ministry of Natural Resources (OMNR). 1996. Forest Resources of Ontario, 1996. Queens Printer for Ontario, Ottawa, ON, 86 p.

Overpeck, J.T., D. Rind and R. Goldberg. 1990. Climate-induced changes in forest disturbance and vegetation. Nature 343: 51–53.

Pan, Y., J.M. Melillo, A.D. McGuire, D.W. Kicklighter, L.F. Pitelka,

K. Hibbard, L.L. Pierce, S.W. Running, D.S. Ojima, W.J. Parton, D.S. Schimel, and other VEMAP members. 1998. Modeled responses of terrestrial ecosystems to elevated atmospheric CO2: a comparison of simulations by the biogeochemistry models of the Vegetation/Ecosystem Modeling and Analysis Project (VEMAP). Oecologia 114: 389–404.

Pastor, J. and W.M. Post. 1986. Influence of climate, soil moisture, and succession on forest carbon and nitrogen cycles. Biogeochem. 2: 3–27.

Pastor, J. and W.M. Post. 1988. Response of northern forests to CO_2-induced climate change. Nature 334: 55–58.

Peters, R.L. 1990. Effects of global warming on forests. For. Ecol. Manage. 35: 13–33.

Peters, R.L. and T.E. Lovejoy. 1992. Global Warming and Biological Diversity. Yale Univ. Press, New Haven, CT.

Piene, H. and K. Van Cleve. 1978. Weight loss of litter and cellulose bags in a thinned white spruce forest in interior Alaska. Can. J. For. Res. 8: 42–46.

Porter, J.H., M.L. Parry and T.R. Carter. 1991. The potential effects of climatic change on agricultural insect species. Agric. For. Meteor. 57: 221–240.

Post, W.M., W.R. Emanuel, P.J. Zinke and A.G. Stangenberger. 1982. Soil carbon pools of world life zones. Nature 298: 156–159.

Prentice, K.C. and I.Y. Fung. 1990. The sensitivity of terrestrial carbon storage to climate change. Nature 346: 48–51.

Price, C. and D. Rind. 1994. Possible implications of global climate change on global lightning distributions and frequencies. J. Geophys. Res. 99: 108–123.

Price, D.T., D.H. Halliwell, M.J. Apps, W.A. Kurz and S.R. Curry. 1997. Comprehensive assessment of carbon stocks and fluxes in a Boreal-Cordilleran forest management unit. Can. J. For. Res. 27: 2005–2016.

Reed, D.D. and P.V. Desanker. 1992. Ecological implications of projected climate change scenarios in forest ecosystems in northern Michigan. Int. J. Biometerol. 36: 99–107.

Reich, P.B., D.F. Grigal, J.D. Aber and S.T. Gower. 1997. Nitrogen mineralization and productivity in 50 hardwood and conifer stands on diverse soils. Ecology 78: 335–347.

Richter, D.D., D. Markewitz, S.E. Trumbore and C.G. Wells. 1999. Rapid accumulation and turnover of soil carbon in a re-establishing forest. Nature 400: 56–58.

Richter, K. 1998. Life cycle assessment of wood products. *In* G.H. Kohlmaier, M. Weber and R.A. Houghton (eds.). Carbon Dioxide Mitigation in Forestry and Wood Industry. pp. 219–248, Springer-Verlag, Berlin.

Roberts, L. 1989. How fast can trees migrate? Science 243: 735–737.

Rollinger, J.L., T.F. Strong and D.F. Grigal. 1997. Forested soil carbon storage in landscapes of the northern Great Lakes region. *In* R. Lal, J.M. Kimble, R.F. Follett and B.A. Stewart (eds.). Management of Carbon Sequestration in Soil. pp. 335–350. CRC Press, Boca Raton, FL.

Rubin, E.S., R.N. Cooper, R.A. Frosch, T.H. Lee, G. Marland, A.H. Rosenfeld and D.D. Stine. 1992. Realistic mitigation options for global warming. Science 257: 148–149, 261–266.

Ryan, M.G., D. Binkley and J.H. Fownes. 1997. Age-related decline in forest productivity: pattern and process. Adv. Ecol. Res. 27: 213–262.

Safranyik, L. 1990. Temperature and insect interactions in western North America. *In* Are Forests the Answer? pp. 166–170. Proc. Soc. Amer. For. Nat. Conv., SAF Pub. 90-02, Bethesda, MD.

Samson, R., P. Girouard, C. Zan, B. Mehdi, R. Martin and J. Henning. 1999. The implications of growing short-rotation tree species for carbon sequestration in Canada. Final report. Joint Forest Sector Table/Sinks Table Afforestation #5, National Climate Change Process Solicitation No: 23103-8-0253/N. 84 p.

Sargent, N.E. 1988. Redistribution of the Canadian boreal forest under a warmed climate. Clim. Bull. 22: 23–34.

Saxe, H., D.S. Ellsworth and J. Heath. 1998. Tree and forest functioning in an enriched CO_2 atmosphere. New Phytol. 139: 395–436.

Schindler, D.W. 1998. A dim future for boreal waters and landscapes. Bioscience 48: 157–164.

Schlamadinger, B. and G. Marland. 1996. Carbon implications of forest management strategies. *In* M.J. Apps and D.T. Price (eds.). Forest Ecosystems, Forest Management and the Global Carbon Cycle. pp. 217–232. NATO ASI Series, Vol. I 40, Springer-Verlag, Berlin.

Schlesinger, W.H. 1977. Carbon balance in terrestrial detritus. Annu. Rev. Ecol. Syst. 8: 51–81.

Schopfhauser, W. 1998. World forests: the area for afforestation and their potential for fossil carbon sequestration and substitution. *In* G.H. Kohlmaier, M. Weber, and R.A. Houghton (eds.). Carbon Dioxide Mitigation in Forestry and Wood Industry. pp. 185–203. Springer-Verlag, Berlin.

Schroeder, P. 1991. Can intensive management increase carbon storage in forest? Environ. Manage. 15: 475–481.

Sedjo, R.A., R.N. Sampson and J. Wisniewski. 1997. Economics of carbon sequestration in forestry. Crit. Rev. Environ. Sci. Technol. 27: 1–364.

Simard, A.J. 1997. National workshop on wildland fire activity. Workshop Report. Natur. Resour. Can., Can. For. Serv., Inf. Rep. ST-X-13. 38 p.

Singh, T. and E.E. Wheaton. 1991. Boreal forest sensitivity to global warming: Implications for forest management in western interior Canada. For. Chron. 67(4): 342–348.

Sittig, M. 1979. Fertilizer industry processes, pollution control, and energy conservation. Noyes Data Corporation, Park Ridge, NJ.

Smith, D.M., B.C. Larson, M.J. Kelty and P.M.S. Ashton. 1997. The Practice of Silviculture: Applied Forest Ecology, Ninth Edition. John Wiley and Sons, Inc., New York, NY.

Smith, J., B. Lavender, H. Auld, D. Broadhurst and T. Bullock. 1998. The Canada Country Study: Climate Impacts and Adaptation. – Volume IV. Adapting to climate variability and change in Ontario. Envrion. Can., Ontario Region. 117 p.

Solomon, A.M. 1986. Transient response of forests to CO_2-induced climate change: simulation modeling experiments in eastern North America. Oecologia 68: 567–579.

Solomon, A.M. and P.J. Bartlein. 1992. Past and future climate change: response by mixed deciduous-coniferous forest ecosystems in northern Michigan. Can. J. For. Res. 22: 1727–1738.

Stewart, R.E., L.L. Gross and B.H. Honkala. 1984. Effects of competing vegetation on forest trees: a bibliography with abstracts. USDA, For. Serv., Gen. Tech. Rep. WO-43. 260 p.

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

Suffling, R. 1995. Can disturbance determine vegetation distribution during climate warming? A boreal test. J. Biogeogr. 22: 501–508.

Thompson, I.D., M.D. Flannigan, B.M. Wotton and R. Suffling. 1998. The effects of climate change on landscape diversity: an example in Ontario's forests. Environ. Monitor. Assess. 49: 213–233.

Turner, D.P., G.J. Koerper, M.E. Harmon and J.J. Lee. 1995. A carbon budget for the forests of the conterminous United States. Ecol. Appl. 5:421–436.

van Kooten, G.C., L.M. Arthur and W.R. Wilson. 1992. Potential to sequester carbon in Canadian forests: some economic considerations. Can. Publ. Policy 18: 127–138.

van Kooten, G.C., E. Krcmar-Nozic, B. Stennes and R. van Gorkom. 1999. Economics of fossil fuel substitution and wood product sinks when trees are planted on agricultural lands in western Canada. Can. J. For. Res. 29: 1669–1678.

Vitousek, P.M. 1994. Beyond global warming: ecology and global change. Ecology 75: 1861–1876.

Vogt, K.A., C.C. Grier and D.J. Vogt. 1986. Production, turnover, and nutrient dynamics of above- and below-ground detritus of world forests. Adv. Ecol. Res. 15: 303–377.

Volney, W.J.A. 1996. Climate change and management of insect defoliators in boreal forest ecosystems. *In* M.J. Apps and D.T. Price (eds.). Forest Ecosystems, Forest Management and the Global Carbon Cycle. pp. 79–87. NATO ASI Series, Vol. I 40, Springer-Verlag, Berlin.

Wallner, W.E. 1987. Factors affecting insect population dynamics: differences between outbreak and non-outbreak species. Ann. Rev. Entomol. 32: 317–340.

Walstad, J.D. and P.J. Kuch. 1987. Forest Vegetation Management for Conifer Production. John Wiley and Sons, New York, NY.

Wang, Z.M., M.J. Lechowicz and C. Potvin. 1995. Responses of black spruce seedlings to simulated present versus future seedbed environments. Can. J. For. Res. 25: 545–554.

Wargo, P.M. and T.C. Harrington. 1991. Host stress and susceptibility. *In* C.G. Shaw and G.A. Kile (eds.). Armillaria Root Disease. USDA, For. Serv., Agric. Handbook No. 691, Washington, DC.

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

Wernick, I.K., P.E. Waggoner and J.H. Ausubel. 1998. Searching for leverage to conserve forests. J. Indust. Ecol. 1:125–145.

Wilson, J.S. and P.J. Baker. 1998. Mitigating fire risk to late-successional forest reserves on the east slope of the Washington Cascade Range, USA. For. Ecol. Manage. 110: 59–75.

Winjum, J.K., S. Brown and B. Schlamadinger. 1998. Forest harvest and wood products: sources and sinks of atmospheric carbon. For. Sci. 44: 272–284.

Winjim, J.K. and P.E. Schroeder. 1997. Forest plantations of the world: their extent, ecological attributes, and carbon storage. Agric. For. Meteorol. 84:153–167.

Wotton, B.M. and M.D. Flannigan. 1993. Length of the fire season in a changing climate. For. Chron. 69: 187–192.

Wullschleger, S.D., J.P. Lynch and G.M. Berntson. 1994. Modeling the belowground response of plants and soil biota to edaphic and climate change – What can we expect to gain? Plant Soil 165: 149–160.

Zerbe, J.I. 1993. Reduction of atmospheric carbon emissions through displacement of fossil fuels. World Res. Rev. 5: 414–423.

Zoltai, S.C. 1988. Wetland environments and classifications. *In* Wetlands of Canada. pp. 1–26. National Wetlands Working Group, Ecological Land Classification Series, No. 24, Sust. Develop. Br., Environ. Can., Ottawa, ON and Polyscience Publications Inc., Montreal QC.

Zoltai, S.C. and P.J. Martikainen. 1996. Estimated extent of forested peatlands and their role in the global carbon cycle. *In* M.J. Apps and D.T. Price. (eds.). Forest Ecosystems, Forest Management and the Global Carbon Cycle. pp. 47–58. NATO ASI Series, Vol. I 40, Springer-Verlag, Berlin.