Chapter 20

Carbon balance and climate change in boreal forests

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> *"The greatest threat of all may be yet to come, in the form of global warming."*
>
> "Great Northern Forest" (1994),
> a film directed by Joseph Viszmeg,
> produced by Albert Karvonen and Jerry Krepakevich,
> Karvonen Films Ltd., and National Film Board of Canada

Introduction

Most scientists agree that the global climate is changing as a consequence of human-caused perturbations to global biogeochemical cycles, especially the carbon cycle (IPCC 2001*a*). In addition, the impacts of these changes are becoming increasingly evident (IPCC 2001*b*). The boreal forest biome is one of the Earth's ecosystems most affected by the changing climate (IPCC 2001*a*; Gitay et al. 2001). The forests in this

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biome also have the potential to either accelerate or retard the progression of climate change through their influence on the global carbon cycle.

This chapter explores the role of the boreal forest in the global carbon cycle, the expected impacts of climate change on the boreal ecosystem, and the effects of various factors (both natural and human) on the carbon balance of the forest. The last part of the chapter discusses the economic and forest management issues that arise when the carbon resources of the forest are considered in light of Kyoto Protocol commitments to reduce greenhouse gas emissions. While the primary focus of this chapter is the challenges (Box 20.1) climate change poses for the Canadian boreal forest, the boreal forest in other regions of the world will also be discussed where appropriate.

Box 20.1. Challenges to sustainable forest management in light of climate change.

Climate change poses many challenges for sustainable forest management in boreal areas. These include determining how to:

- manage carbon pools and fluxes, as well as the fibre supply from trees and stands;
- minimize use of fossil fuels;
- control disturbance, growth, and decomposition processes in an economically and ecologically feasible way;
- exploit opportunities for carbon sequestration and associated credits;
- maintain the ecological integrity of the forest in the face of increasing climate change and other global change stresses; and
- inventory forest ecosystem carbon stocks and to estimate carbon sources and sinks.

Forests and climate

Forests are highly influenced by climate, which determines their distribution, structure and composition, and much of their ecological function. Climate also affects forests indirectly through its impact on disturbance regimes such as those due to fire, insects, diseases, and windstorms. In turn, forests have an impact on global climate through their influence on surface roughness, albedo, hydrological cycles, and the carbon cycle. The influence of the boreal forest on climate through its effect on the global carbon cycle has been receiving increasing attention in recent years because of escalating concern about climate change. The role of carbon in climate change, the importance of the boreal forest in the global carbon cycle, and the expected impacts of climate change on the boreal forest are explored below.

The role of carbon in global climate change

Scientific background

The Earth's climate is determined in large part by the proportion of solar energy retained by the Earth's atmosphere. Certain gases (called greenhouse gases or GHGs) in the atmosphere allow short-wavelength solar energy (visible light) to pass through to the Earth but absorb outgoing infrared (radiant heat) wavelengths, thus making the Earth warmer than it would otherwise be. Without this greenhouse effect the Earth would be uninhabitable for most existing lifeforms, including humans. Naturally occurring GHGs include water vapour (H_2O) and various trace gases such as carbon dioxide (CO_2), methane (CH_4), ozone (O_3), and nitrous oxide (N_2O). Synthetic GHGs include chlorofluorocarbons (CFCs). The concentrations of GHGs in the atmosphere are largely regulated by global hydrological and biogeochemical cycles, including the carbon cycle.

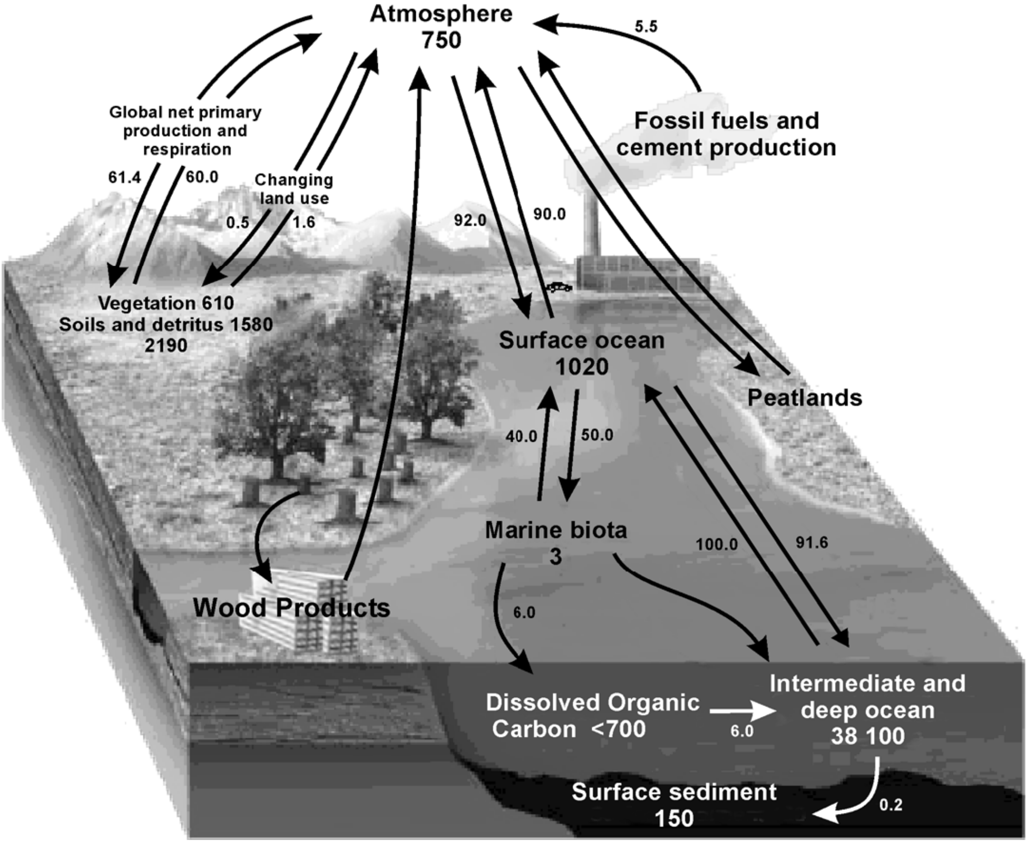
Human activities such as burning fossil fuels, production of cement, and changes in land use have perturbed the carbon cycle, which has resulted in elevated levels of atmospheric CO_2, thus enhancing the greenhouse effect and causing global warming at an unprecedented rate (IPCC 2001*a*). It is estimated that during the period from 1850 to 1998, 405 Pg (1 Pg = 10^{15} g, 10^{12} kg, or 10^9 Mg or tonnes) of carbon was emitted to the atmosphere as a result of these activities (IPCC 2000). This has resulted in a 28% increase in atmospheric CO_2 concentrations (IPCC 2000), which is believed to be a primary cause of the observed 0.6°C increase in global temperature in the 20th century (IPCC 2001*a*). The climatic changes are, however, not uniformly distributed: the changes in solar insolation cause changes in atmospheric circulation patterns, resulting in changes in regional weather patterns. Some regions, especially northern and mid-continental landmasses, undergo larger climate changes than others.

Carbon thus plays a vital role in determining global climate. The global carbon cycle and the exchange of carbon between the atmosphere and various natural and anthropogenic components are illustrated in Fig. 20.1. Carbon is exchanged between terrestrial ecosystems and the atmosphere through photosynthesis, respiration, decomposition, and combustion. Boreal forests and their associated peatlands represent the largest terrestrial reservoir of carbon (IPCC 2000), as well as being located in a region especially sensitive to climate change. The boreal biome therefore plays a critical role in the global carbon cycle and has the capacity for either accelerating or slowing climate change to some degree, depending on whether the forest acts as a net source or a net sink of carbon. This source or sink status is, however, not a static characteristic of the ecosystem, but can change over time as a result of changes in forest age-class structure, disturbance regime, and resource use (Kurz and Apps 1999; Kauppi et al. 2001).

Projected future climates of boreal regions

Global climate is expected to warm by an average of 1.4–5.8°C by 2100, but the temperature increase in the boreal region and other high latitude biomes in the Northern Hemisphere is predicted to be more than 40% higher than this (IPCC 2001*a*). Changes in precipitation are more difficult to predict, but several models suggest regional changes in summer and winter precipitation of ±20% for the boreal region (Kirschbaum and Fishlin 1996; Amiro et al. 2001*a*). For example, precipitation is expected to decrease by 20% in northern Alberta, Saskatchewan, and Manitoba (Amiro et al. 2001*a*; Flannigan et al. 2001), while parts of eastern Canada (Flannigan et al. 2001; Amiro et al. 2001*a*) and Fennoscandia (IPCC 2001*b*) are expected to experience an increase in precipitation. Higher temperatures will result in higher evaporation rates, and hence

Fig. 20.1. Overview of the global carbon cycle, showing the stocks and fluxes of C (Pg) between various components of the biosphere (courtesy of Canadian Forest Service).



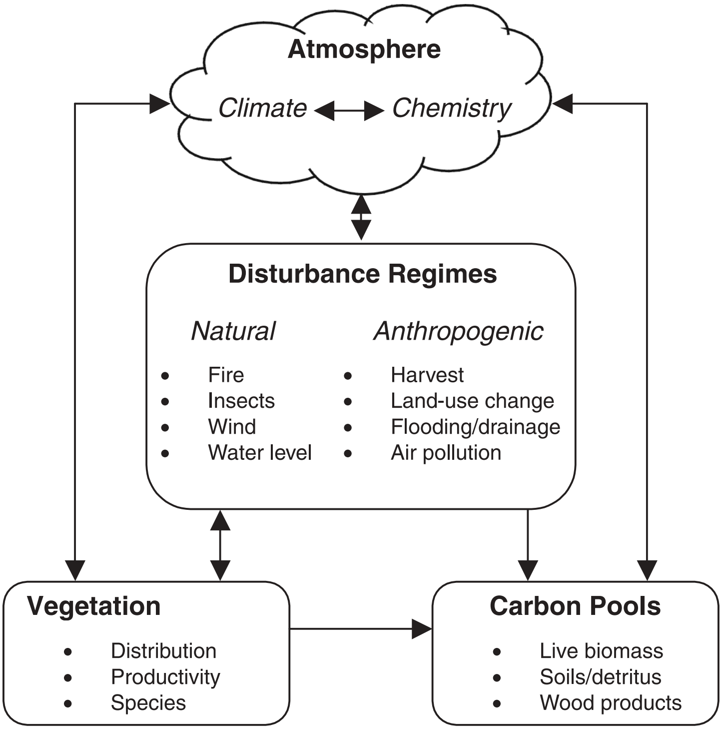
soils in some boreal regions are expected to be drier in the summer (Flannigan et al. 1998; Amiro et al. 2001*a*). The frequency and severity of storm events is also expected to increase (IPCC 2001*a*).

Boreal forests have experienced large climate swings in the past. During geologically recent Ice Ages, almost the entire present range of the North American boreal forests was glaciated, so that most of these forests have occupied their present range for only a few hundred generations of trees. Boreal forests therefore are composed mainly of species that are able to spread relatively quickly over previously unvegetated or recently vegetated terrain. Even these relatively rapid migration rates may not be fast enough to keep pace with the coming climate change, however, which is predicted to occur at a rate at least 10 times faster than the warming since the end of the last Ice Age (Schneider 1995).

The effects of climate change and other global changes on forests

Climate change is expected to affect both the distribution and the character of the boreal forest through changes in temperature, precipitation, and natural disturbance patterns (IPCC 2001*b*). These impacts on the forest are not entirely separable from the effects of

Fig. 20.2. Feedbacks between the atmosphere and various components of the boreal forest (modified from Bhatti et al. 2001).



other global changes, such as increases in CO_2, NO_x, and O_3 levels, and anthropogenic pressures which may be enhanced by climate change. Figure 20.2 illustrates the web of interaction among climate, vegetation, disturbance regimes, and carbon pools in the boreal forest. Various components of this web are discussed in the following sections.

Water and energy exchange

Climate change has the potential to influence the dynamics of water and energy exchange between boreal ecosystems and the atmosphere in several ways. At the forest–tundra boundary, sharp discontinuities are hypothesized to have a strong influence on temperature and precipitation (Eugster et al. 2000). Snow-covered tundra and boreal peatlands have a much higher albedo, absorb less radiation, and warm the atmosphere less than the forested areas (Chapin et al. 2000). Consequently, an expansion of the boreal forest into regions now occupied by tundra and peatland (see below) has the potential to reduce albedo and increase spring energy absorption, thereby enhancing atmospheric warming (Bonan et al. 1992; Chapin et al. 2000). Other factors that enhance atmospheric warming in the boreal forest include earlier snowmelt (which reduces springtime albedo), and increased ground vegetation cover (which reduces evapo-transpiration). Deciduous forests have twice the albedo and 50–80% higher evapo-transpiration than coniferous forests during the growing season (Baldocchi et al. 2000). Changes in the disturbance regime that increase the proportion of deciduous forests have the potential to reduce spring energy absorption and act as a negative feedback to atmospheric warming (Chapin et al. 2000; Eugster et al. 2000). With frequent fire, young boreal forest stands may have a ratio of sensible to latent heat that is 8 times higher than that of mature stands (Schulze et al. 1999).

Forest distribution and character

Both simple and more complex models agree that the ranges of major forest types around the world are likely to shift dramatically with future global climate change. Models using modern correlations between climate and vegetation have projected the stable ranges that might be achieved several hundred or thousand years from now. In many regions, the change is not simply from one forest type to another, but rather from forest to grassland or even desert in some regions, and from tundra to forest in others (Rizzo and Wiken 1992). The predicted stable ranges are, however, in many ways less of a concern than are the transitional displacements. Harder to predict than eventual stable ranges (should they actually stabilize), the transitional state may include significant areas of essentially dead forest, where the existing vegetation is not able to cope with changed climatic conditions, and a suitably adapted natural forest type has not yet had time to migrate and establish itself.

Temperature-limited tree lines are expected to advance, either uphill in mountain regions, or into tundra regions in the Arctic (IPCC 2001*b*), although this potential is not likely to be realized in the next 100–200 years. At the same time, it has been postulated that climate warming will cause many dry forest regions (limited by water availability rather than temperature) to give way to grassland or more open savanna as a consequence of regeneration failure (Hogg and Hurdle 1995; Lenihan and Neilson 1995; Hogg 1997). Temperature is not the sole control on species distribution, and change in temperature cannot be considered in isolation. Other factors, including soil characterization, nutrient availability, and disturbance regimes may prove to be more important in controlling future ecosystem dynamics. For example, the southern limit of boreal forests appears to be influenced more by interspecific competition and moisture conditions than by temperature tolerance (Loehle 1998). The distribution of trembling aspen (*Populus tremuloides* Michx.) in western Canada is largely controlled by moisture conditions (Hogg 1999). A decrease in water availability with climate change may therefore alter the southern boundary of the boreal forest and the distribution of aspen in some regions. Inertial effects may slow the loss of forest in dry regions, although many dry regions also experience human population pressures tending toward deforestation. For example, conifer density will likely decline in northeastern China, where climate warming may reduce the area of coniferous forest substantially, while elsewhere in Asia forests will be under increasing human pressures that will mask the effects of global warming (IPCC 2001*b*).

Many of the changes in forest distribution will not be brought about directly through the impact of climate change on individual trees, but rather through the effect of climate change on disturbance regimes and subsequent regeneration (Flannigan et al. 2000, 2001). The structure, pattern, and ecosystem processes of forests in many regions of the world are highly sensitive to changes in disturbance regimes (see Chap. 9). In north-

Fig. 20.3. Leaning dead trees are a good indicator of permafrost collapse. Permafrost melt can be initiated by climate warming and (or) fire.

(*a*)

Photo by Steve Zoltai, courtesy of Canadian Forest Service

(*b*)

Photo by Merritt Turetsky, courtesy of U.S. Geological Survey

eastern North America, the relative dominance of various tree species is largely determined by the mean fire return interval, which in turn is influenced by the vegetation, creating an interacting spiral (Flannigan and Woodward 1994). This feedback loop may result in meta-stable vegetation formations, sometimes in one state (e.g., grass-dominated), sometimes in another (e.g., tree-dominated). These highly non-linear feedback effects may result in vegetation changes more rapid than would be predicted from climate change alone.

One of the factors affecting regional response to rapid climate change is the existence of barriers to the immigration of species better adapted to the new climate in a given area. For instance, the climate north of the Great Lakes may become suitable for more thermophilous forest types than presently occur there, but the barrier formed by the lakes may slow or prevent suitable species from spreading northward. It has also been suggested that tundra afforestation may be slowed by the need for soil development (Rizzo and Wiken 1992). Most tree-line species, however, are adapted to growing on sites with minimal soils, since they are the same species that colonized much of the boreal region after deglaciation. Permafrost and changes in the active soil layer with warming add significant complexity to the tree-line response in much of Siberian Russia, Canada, and Alaska.

In some mountainous regions, altitudinally rising climatic zones may displace some ecozones from the tops of the mountains, which will lead to extensive extirpations. At the same time, valleys may develop climatic conditions suitable for species not currently present, allowing an opportunity for invasive species or causing temporary loss of forest cover. Such a situation is likely to lead to a large loss of biodiversity, as the first species suited to the new climate rapidly become dominant and resist the invasion of later immigrants. Over time, this inertial barrier may be overcome, but not if the climate continues to change and transient responses continue to dominate.

Invasive species may also increase in non-mountainous areas because of increased fragmentation (see Chap. 12). While there is a vigorous debate on the role of biodiversity in community resistance to invasive species, there is also general agreement that forest fragmentation (from land-use change, direct climate change effects, and possible increased disturbances due to warming) will likely lead to increased opportunities for invasive and exotic species. Additionally, changing climate zones may further encourage plant and insect migrations (Gitay et al. 2001), along with intercontinental transfers through trade. Invasive insects may be further encouraged where trees are weakened by drought or other climate change effects, so some forests may become less resistant to invasive insects and native insects alike (see Chap. 8 and below).

Forest fragmentation will also lead to fragmentation of wildlife habitat, and possible extinctions. Anthropogenic forest fragmentation was a major factor in the extinction of the passenger pigeon in eastern North America at the beginning of the 20th century (Shorger 1972), and habitat fragmentation due to climate change has been proposed as a factor in the extinction of mammoths, sloths, and other large fauna after deglaciation (Dixon and Dixon 1991).

At high latitudes and altitudes, permafrost melt-out will exacerbate and complicate ecosystem responses. Melting permafrost leaves scars on the landscape known as thermokarst depressions; permafrost collapse in peatlands creates features called internal lawns (Vitt et al. 2000). These depressions are often waterlogged and usually involve collapse of soil surfaces. Paludification and soil slumping in turn lead to tree mortality and leaning trees, referred to as a "drunken forest" (Fig. 20.3). The eventual result is frequently a wet sedge meadow embedded in the forest.

Natural disturbances

Fire — In Canada, climate change is expected to contribute to a significant increase in forest fire activity (Fig. 20.4) particularly in western and central Canada (Flannigan et al. 1998). This will occur through a longer fire season, larger and more intense fires resulting in part from increased periods of drought, and an increase in both natural and anthropogenic fire ignitions (Stocks et al. 2000). The direct impact of climate change on

Fig. 20.4. Climate change may increase forest fire activity in some boreal regions.

Photo by Brian Amiro, courtesy of Canadian Forest Service

tree mortality may increase fuel loads in some regions. Climate change thus poses two major problems for fire research: predicting the impact of climate change and consequent vegetation changes on the fire regimes; and predicting the impact of increased fire activity on the vegetation and soil and, through their carbon storage, on global climate.

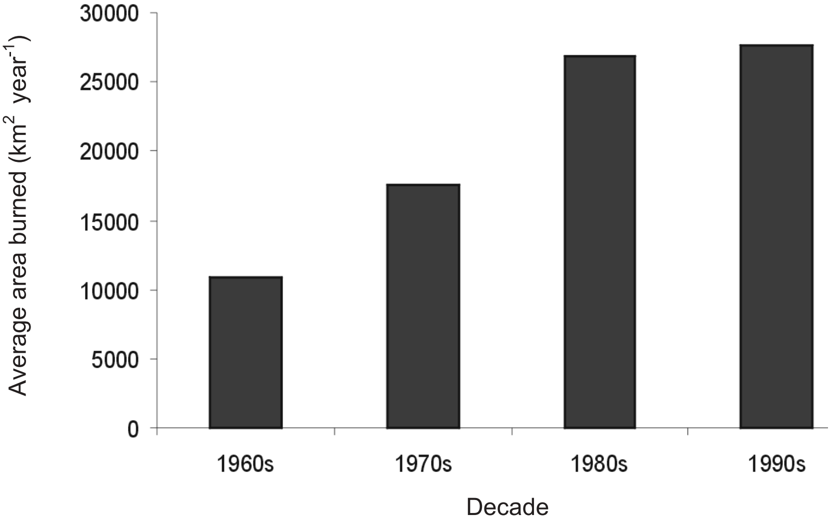
Forest fires affect the global carbon cycle in several ways (Kasischke 2000). First, the fire directly releases large quantities of carbon into the atmosphere through combustion of plant material and surface soil organic matter. Secondly, carbon is released from the decomposition of fire-killed vegetation over time. Thirdly, for several years or decades after a fire, the vegetation on newly burned sites may not fix as much carbon from the atmosphere as did the pre-fire vegetation. In addition it can take many decades before the ecosystem carbon stocks (in vegetation, on the forest floor, and in the soils) return to their pre-fire levels (Apps et al. 2000), and recover the carbon released during and after the fire event. Fires are thus an important part of the global carbon cycle, with increased fire frequency generally causing a net reduction in sequestered carbon stocks (Kurz et al. 1995*b*). Forest fires may also lead to replacement of one vegetation type with another. Frequent fires in some areas are responsible for creating essentially permanent openings in the forest and may be a major factor in excluding conifers from the Aspen Parkland ecoregion (Hogg 1997) and white cedar (*Thuja occidentalis*) from some sites in eastern Canada (Larocque et al. 2003). At the same time, fire may facilitate the adaptation of the forest to climate change by eliminating vegetation that is no longer optimally adapted to the new climate and permitting the development of new, better adapted vegetation communities (Arseneault and Payette 1992).

Fire itself responds strongly to climate change. The annual area burned will likely increase, perhaps by as much as 50% over the coming decades, in response to global warming (Flannigan and van Wagner 1991; Stocks et al. 1998). This increase is not likely to be evenly distributed. For example, parts of eastern Canada may actually experience a decrease in fire, while western and central Canada experiences a significant increase (Flannigan et al. 2001). There is some evidence to suggest the fire regime in Canada may already be responding to climate change (Amiro et al. 2001*a*; Stocks et al. 2003). Since the 1960s, the average annual area burned has increased from about 1.5 million ha to nearly 3 million ha (Fig. 20.5). A small part of this increase may be due to improved detection capacity, but much of the difference reflects the influence of new weather patterns (possibly due to climate change) as well as other factors, such as aging of the forest and increased human activities in formerly remote areas.

Insects and pathogens — Increases in tree mortality associated with insects and disease may be expected under a changing climate. Existing forests in the process of adapting to rapid global change, whether through growth stimulation or degradation, are under stress. Such conditions tend to increase their susceptibility to insect and disease attack. In addition to these changes in host conditions, both insect population and disease cycles may be directly affected by climate change. The incidence of exotic insects and disease can be expected to increase as climatic conditions make migration and survivorship more probable. Human transportation routes and activities exacerbate this problem. Insect disturbances are expected to increase with climate change because of longer growing seasons, increased forest stress, and less severe winters (Fleming and Volney 1995). Climate change may also affect insect diversity, allowing more exotic species to survive in the northern forests (Gitay et al. 2001).

Many uncertainties remain regarding the character, magnitude, and rates of future climate change and its impact on insect disturbances. However, it is generally expected that insect outbreaks will become increasingly more intense with climate change (Gitay et al. 2001) as a consequence of both atmospheric warming and CO_2 enrichment (Fleming and Volney 1995). In general, current forecasts of forest insect responses to climate

Fig. 20. 5. Area burned in Canada. Significantly more area was burned in the 1980s and 1990s than in the previous two decades (data compiled from Stocks et al. 2003).



warming, which are based on historical relationships between outbreak patterns and climate, would suggest:

- more frequent outbreaks of
    - mountain pine beetle, *Dendroctonus ponderosae* (Thompson and Shrimpton 1984),
    - spruce budworm, *Choristoneura fumiferana* (Mattson and Haack 1987),
    - eastern hemlock looper, *Lambdina fiscellaria fiscellaria* (Carroll et al. 1995);
- longer outbreaks of
    - forest tent caterpillar, *Malacosoma disstria* (Roland et al. 1998), and
    - spruce budworm (Cerezke and Volney 1995); and
- range shifts northward and to higher elevations by
    - spruce budworm (Williams and Liebhold 1997),
    - gypsy moth, *Lymantria dispar*, and
    - white pine weevil, *Pissodes strobi* (Sieben et al. 1994).

Recent evidence suggests that some insect populations may already be responding to climate change. For example, large outbreaks of defoliating insects have been observed in recent years at much higher latitudes than previously (Brandt et al. 1996). Changes in seasonality (especially warmer and shorter winters) have also been implicated in more frequent outbreaks of bark beetle infestations in southern Alaska and mountain pine beetle in British Columbia. A recent survey showed that the pine beetle damaged approximately 1 million ha in British Columbia in 2002 and 5.7 million ha are currently at risk (Safranyik and Linton 2002). Scientists partially attribute the epidemic to global warming. Pine beetles usually have a 2-year cycle, but warmer temperatures can cause them to complete their cycle and breed in 1 year (Safranyik and Linton 2002). Warmer temperatures also reduce insect die-off, as fewer insects are killed by very cold weather in the late fall and early spring.

Forest management practices, such as the spraying of pesticides to control insects, can sometimes curtail the pattern, rate, and magnitude of small-scale insect disturbances. However, prediction and prevention of outbreaks is sometimes difficult, and managing infestation over large or remote areas is usually not practical (IPCC 2000).

In addition to affecting insect outbreaks, the generally warmer temperatures and milder winters expected with climate change may also increase the survival of existing pathogens and allow new pathogens to move into the boreal zone. For example, *Armillaria* root disease is a globally distributed fungal disease with optimum growth at 25°C; boreal and temperate forests may therefore experience increased *Armillaria*-caused volume loss with global warming (Mallett and Volney 1999). In areas where drought is expected to increase, there may be a greater incidence of canker diseases in some tree species owing to decreased bark moisture content (Bloomberg 1962; IPCC 2001*b*).

Permafrost melting — Permafrost is an important component of many northern forest ecosystems, and has profound influences upon hydrology, vegetation, and carbon storage. It is discontinuous with distribution being heavily influenced by autogenic factors on a local scale. However, on a more regional scale allogenic factors such as climate also play a major role in the presence or absence of permafrost. The influence of autogenic controls can be so strong that there can be permafrost thaw co-existing with permafrost aggradation within one peatland even if the climate is stable (Zoltai 1972).

Warmer temperatures have resulted in increased permafrost melt in some regions of the boreal forest. For example, air temperatures have warmed significantly in the Mackenzie Valley, N.W.T., ranging from 60 to 64°N, over the past 5 decades (Robinson et al. 2001). In the northern region of the valley, permafrost is still dominant and thaw is restricted to relatively small individual bowl-shaped features. However, over the past 50 years, lateral thaw has increased the size of existing thaw features and a significant number of new features have developed. In the southern region of the valley, individual thaw features have coalesced, creating an integrated drainage network and in many cases isolating the remaining permafrost features (Robinson et al. 2001). Recent thaw in more southern parts of the discontinuous permafrost zone of the boreal forest has also been noted at sites in Manitoba (Thie 1974), and the other prairie provinces (Vitt et al. 2000; Beilman et al. 2001).

In boreal forests, permafrost affects soil moisture because it inhibits permeability and affects the availability of water to the ecosystem during seasonal thaw. Soils with shallow permafrost may be particularly sensitive to climate change because of the role that moisture plays in these ecosystems. For example, shifts toward wetter or cooler conditions might suppress fires and enhance carbon storage. Shifts toward drier summers might favour fire activity and enhance fire emissions. While these climatic trends apply to all systems, forest ecosystems with permafrost soils are at greater risk. In years that favour deep thaw and drying of these deep, humic horizons, fire can tap into these vast quantities of carbon-rich fuels (Harden et al. 2000).

Other natural disturbances (storms, floods, landslides) — Forest damage due to windstorms, ice storms, flooding, and landslides can also be expected to increase with climate change (IPCC 2001*a*). Although insect and fire disturbances will probably be more widespread, forest damage due to storms could be substantial in some regions. For example, the January 1998 ice storm (Fig. 20.6) affected 50 000 km^2 of forest in southeastern Ontario and western Quebec and caused moderate to severe damage to 15 000 km^2 of forest (NRCan 1999*a*). In Europe during 1999, windthrow caused by three storms over a 3-day period resulted in losses of 193 × 106 m^3 of timber, equivalent to 2 years of harvest (Updegraff-ECE/FAO 2000). Such storms may become more frequent in association with milder winters in the future (Cohen and Miller 2001). In the Great Lakes area, windthrow is a major disturbance agent affecting thousands of hectares with both immediate and long-term impacts. Most boreal tree species are shallow-rooted and hence vulnerable to windthrow. Factors such as tree height, whether the tree is alive or dead, and stand density determine whether a tree is just snapped off or is completely uprooted. In boreal cordilleran areas, flooding and landslides due to intense precipitation events could increase in frequency with climate change (Gitay et al. 2001).

Climate change-induced or enhanced anthropogenic pressures

In many regions, climate change impacts on forest distribution will be heavily mediated by its impacts on agriculture. New forest clearance for agricultural expansion into areas that are presently too cold for agriculture may be a major force in modifying forest distributions. At the same time, some marginal agricultural land may be taken out of production and allowed to revert to forest through natural regeneration or planting. This

Fig. 20.6. Trees damaged in 1998 ice storm. Ice storms may become more frequent with climate change.

Photo by Genevieve Zevort

latter trend may be enhanced by government policies favouring afforestation for carbon sequestration purposes (van Kooten et al. 2002). In some regions of the world, however, given the expanding population and consequent demand for agricultural land, the net pressure will be toward deforestation rather than afforestation. For example, one model that integrates the effects of climate change and land-use change suggests a 50% decline in forest cover in parts of Asia by 2050 (IPCC 2001*b*). The trend towards deforestation will be reinforced by the tendency of climate warming to make forested land suitable for agriculture, by increasing the length of the growing season in cold regions, and its tendency to make current agricultural land unsuitable for agriculture through increasing drought. However, in some areas this trend will likely be dampened by the fact that many subarctic lands have soils unsuitable for agriculture, not just unsuitable climate.

Combined with land-use changes, climate change may increase pressures to grow fibre in managed plantations. Because most managed plantations tend to focus on a single (or limited number of) tree species and comprise even-aged stands, they have had a significant effect on biodiversity in some regions (Harris 1984). If the pressures to increase fibre production rise with climate change, as expected, forward-thinking management practices would be required to mediate these effects by, for example, deliberately employing mixed species management and variable cutting patterns, and by planting species better suited to expected future conditions. Some of these options are explored from several perspectives in Chap. 13.

CO_2 fertilization

Under experimental conditions, some species increase their rates of photosynthesis with increases in CO_2 (Farquhar et al. 1980; Wullschlenger et al. 1995; Isebrands et al. 2000; Houghton et al. 2000), partly because of greater water-use efficiency (Field et al. 1995; Farquhar 1997; Körner 2000). This effect is known as CO_2 fertilization. It is hypothesized that CO_2 fertilization may account for up to 33% of the CO_2 absorbed annually through photosynthesis by terrestrial plants (Norby et al. 1999). The CO_2 fertilization effect may enhance the growth of some tree species and forest ecosystems, allowing them to remove more carbon from the atmosphere. Whether the enhancement of photosynthesis by elevated CO_2 actually results in net removal of carbon from the atmosphere at the ecosystem level is a subject of intense debate (e.g., Luo et al. 2003). Recent work based on forest inventory data indicate that the net effect on carbon stocks is very much less than that suggested above, accounting for less than a few percent increase in accumulated carbon in vegetation (Caspersen et al. 2000).

Many of the experimental studies on elevated CO_2 response have been conducted on immature trees, often in growth chambers, under conditions not otherwise limiting plant growth and not measuring the effects of changes in ambient CO_2 concentration that would occur in forests (Curtis and Wang 1998; Norby et al. 1999). Several field experiments are currently under way which employ free air CO_2 enrichment (FACE) technology by which the CO_2 (and other gases) around growing plants has been modified to simulate future levels of these gases under climate change (Curtis et al. 2000; Luo et al. 2003). These experiments, however, have not been running long enough to determine what the long-term effects of elevated CO_2 levels might be once canopy closure is reached (IPCC 2000). While the response of mature forests to increases in atmospheric CO_2 concentration has not been demonstrated experimentally, it may be different from that of individual trees and different from that of young forests (Körner and Bazzaz 1996; Curtis and Wang 1998; Norby et al. 1999; Casperson et al. 2000).

Some scientists have hypothesized that Canada's forest net primary productivity (NPP) may be increasing, and that this increase may be due in part to CO_2 fertilization (Chen et al. 2000). However, inventory measurements in North America indicate that CO_2 fertilization has not appreciably affected carbon accumulation rates in forest biomass over the past century (Casperson et al. 2000). Forest age-class dynamics, land-use change, and alterations in natural disturbance patterns have a much larger influence than CO_2 fertilization on forest growth in this region (Kurz and Apps 1999; Casperson et al. 2000). There is a growing consensus in the scientific community that any CO_2 fertilization effects that may exist are expected to saturate (that is, their contribution to continued net CO_2 removals will go to zero) over the next 100 years or so (Bazzaz et al. 1996; IPCC 2000; Schimel et al. 2001). This occurs because increases in CO_2 levels stimulate increases in photosynthesis at a diminishing rate, while increases in temperature stimulate increases in respiration at an exponential rate (IPCC 2000).

NO_x fertilization and ozone

Nitrogen deposition — The amount of available nitrogen in a given forest ecosystem plays an important role in plant biodiversity, the population dynamics of species in the associated food web, and ecological processes such as plant productivity, winter hardiness, and biogeochemical cycling (Vitousek et al. 1997). In boreal forest ecosystems, nitrogen is a limiting factor, as most of it occurs in forms that cannot be readily used by most plants.

Human activities have increased the supply of nitrogen in some regions of the boreal forest. It has been suggested that increased nitrogen deposition (due to NO_x atmospheric pollution; see Chap. 18) may temporarily enhance forest carbon sequestration in nitrogen-limited ecosystems, leading to a short-term carbon gain in NPP (Nadelhoffer et al. 1999). However, if there is a positive effect on forest growth from nitrogen deposition in boreal forests, it will likely be negated in the medium term as other factors become limiting, such as other nutrients and water (Olsen et al. 2001). In North America, nitrogen deposition has not appreciably affected carbon accumulation rates at the landscape level (Casperson et al. 2000). On the contrary, evidence shows that excess deposition has had harmful effects, at the stand level, on forested and aquatic ecosystems (Schindler 1998; Box 20.2).

Tropospheric ozone — Scientists believe that up to 49% of global forests (17 × 106 km^2) will be exposed to damaging levels of tropospheric ozone (O_3) by 2100 (Fowler et al. 1999). O_3 (smog) is a secondary photochemical oxidant formed from reactions between primary pollutants (nitrogen oxides and volatile organic compounds) and

Box 20.2. Nitrogen deposition from various sources of air pollution can have a number of detrimental effects on ecosystem functions and environmental quality.

It has been shown, for example, that excess nitrogen:

- is a major contributor to photochemical smog and acidification of fog, which alters ecosystem processes and health (Lovett 1994);
- results in losses of soil nutrients, such as calcium and potassium, thus reducing long-term soil fertility (Ouimet and Camire 1995; Likens et al. 1996);
- acidifies the soil, which may have a long-term negative impact on forest growth in some regions (Likens et al. 1996);
- is exported from forested watersheds, where it contributes to episodic acidification and eutrophication of aquatic ecosystems (Schindler 1998);
- may reduce the ability of plants to process CO_2 assimilated from the atmosphere;
- may reduce root to shoot biomass ratios and thus (1) increase plant sensitivity to drought, counteracting possible water-use efficiency gains from CO_2 fertilization (Nadelhoffer et al. 1999), and (2) increase the likelihood of nutrient deficiency due to smaller root mass (Townsend et al. 1996); and
- can accelerate losses of existing biological diversity, especially among plants that are adapted to low-nitrogen soils, by increased competition from plants more responsive to higher nitrogen levels (grasses), and subsequently affect the animals and microbes that depend on them (Lovett 1994).

sunlight. These primary pollutants are produced by activities such as the burning of fossil fuel and emissions from transportation. Ozone is also a greenhouse gas, trapping outgoing heat radiation and radiating it back to Earth, and is estimated to be responsible for up to 30% of radiative forcing (Stevenson et al. 1998). Ozone is the most pervasive and toxic gas to plants in the lower atmosphere.

Although the precursors of nitrogen oxide pollutants are primarily created in urban areas, air masses carry them over long distances, resulting in significant tropospheric ozone levels over regional and continental scales. The highest levels of O_3 are produced on warm sunny days with relatively still air, in mid to late summer, during the late morning and early afternoon, particularly in areas with thermal inversions.

Annual mean ground-level O_3 concentrations in Canada are increasing, particularly in urban areas (Munn and Maarouf 1997). At least 2×10^6 ha of Canada's productive eastern forest is exposed annually to damaging levels (McLaughlin and Percy 1999). Exposure of western forests to O_3 is difficult to estimate because of a lack of ground-level monitoring data, but some northwestern forest ecosystems will likely be more exposed through significant industrial expansion in these areas. Ozone can harm forest ecosystems by impairing tree physiology, in particular by decreasing the rate of photosynthesis in some species and altering carbohydrate allocation patterns in others (Percy et al. 1999). With respect to the latter, carbon transfer is commonly decreased to the roots and increased to the shoots, which makes trees more vulnerable to drought, nutrient deficiencies, and winter damage.

Results from FACE studies indicate that while increased CO_2 alone results in an increase in growth in some tree species, the increases are often negated by the effects of tropospheric ozone (Isebrands et al. 2000). For example, Isebrands et al. (2000) reported negative responses in aspen (*Populus tremuloides* Michx.) and birch (*Betula papyrifera* Marsh.) aboveground estimated stem volumes relative to the controls after 3 years of fumigation with O_3 or $O_3 + CO_2$. A stimulation of 20–30% with CO_2 alone was also completely offset by O_3. While experimental studies have shown reduced growth rates in some forest species exposed to O_3 (Isebrands et al. 2000), there is no evidence that changes in O_3 levels are resulting in any significant changes (either negative or positive) in forest growth rates at the landscape level. Any effects are likely to be area-specific and occur against a longer term background of climate and forest change.

The ozone fumigation facility at the Canadian Forest Service's Atlantic Forestry Centre is the only facility in Canada currently investigating the combined effects of human-induced stresses involving O_3. These studies on birch species (*Betula* spp.) include interactions between summer O_3 exposure and extended winter thaws and (or) late frosts (likely to increase under global warming), and interactions between excess nitrogen fertilization and extended winter thaws (Percy et al. 1999).

The "natural" carbon budget of forests

The boreal forest biome contains approximately 700 Pg of carbon (Apps et al. 1993*a*), a significant proportion (25%) of the global terrestrial carbon pool, estimated at 2500 Pg (IPCC 2000). The net amount of carbon stored in the forest fluctuates naturally over time. The following sections give an overview of current estimates of carbon stocks and fluxes in the boreal region, the natural factors affecting these stocks, and the ways in which climate change is expected to affect the carbon budget of the boreal forest.

Current forest carbon stocks

The estimated sizes of various carbon pools in the boreal forests of Canada, Alaska, Russia, and Scandinavia are summarized in Table 20.1. There is much uncertainty underlying the carbon estimates for the peatland component because of the lack of information on the true extent and depth of peatlands and the unknown degree of overlap between peatland and forest inventories (Apps et al. 1993*a*). The carbon stocks and fluxes of various carbon pools are discussed in greater detail in the following sections.

Above- and below-ground stocks and dead organic matter

According to Bhatti and Apps (2000), aboveground biomass in boreal forests varies between 22 and 187 Mg ha^{-1} (1 Mg = 10^6g). In general, drier sites contain less biomass whereas wetter sites contain more biomass. The higher biomass under poorly drained conditions is associated with the presence of many mature black spruce (*Picea mariana* (P.Mill.) B.S.P.) stands that are long-lived because wet sites are less susceptible to fire disturbance (Bhatti et al. 2002*a*). Conversion of biomass to carbon indicates that carbon storage in these boreal forests ranges from 11 to 97 Mg C ha^{-1}. These estimates are consistent with estimates of 5–54 Mg C ha^{-1} (average 24 Mg C ha^{-1}) reported by Simpson et al. (1993) from direct measurements of western Canada's boreal forest. On the basis of point measurements, soil survey data sets, and modelling, Bhatti et al. (2002*b*) estimated that belowground carbon stocks for upland forest soils range from 14 to 78 Mg C ha^{-1} for the surface layer and from 62 to 274 Mg C ha^{-1} for the total soil column.

Table 20.1. Boreal forest biome carbon pools and fluxes, based on data available prior to 1993 (compiled from Apps et al. 1993*a*; Weber and Flannigan 1997).

	Canada	Alaska	Russia	Scandinavia
Area (Mha)				
Boreal forest	304	52	760	61
Peatland	89	11	136	20
Pools (Pg C)				
Plant biomass[a]	8	2	46	2
Plant detritus	—[b]	1	31	NA[e]
Forest soil	65	10	100	NA
Peat	113	17	272	13
Fluxes (Tg C/year)[c]				
Boreal forest[d]	62	6	493	43
Peatland	25	3	11	5

[a] Above- and below-ground live biomass.

[b] Plant detritus estimates are included in the soil carbon pool.

[c] Flux values represent net transfers from the atmosphere.

[d] Includes biomass and soil C pool dynamics.

[e] NA, not available.

These soil carbon estimates are comparable to those reported for global boreal forests (111–190 Mg C ha^{-1}) by Post et al. (1982), and for North American boreal forests (135–195 Mg C ha^{-1}) by Pastor and Post (1988).

Aboveground annual biomass productivity, NPP, compiled by Gower et al. (1997) for boreal forest pine (*Pinus* spp.) stands range between 2.3 and 7.0 Mg ha^{-1} $year^{-1}$ with an average of 4.2 Mg ha^{-1} $year^{-1}$. The average aboveground NPP of three different stand types (mature aspen, black spruce, and jack pine [*Pinus banksiana* Lamb.]) at BOREAS (Boreal Ecosystem–Atmosphere Study) research sites was estimated at 1.7 Mg C ha^{-1} $year^{-1}$ in 1993 and 1.8 Mg C ha^{-1} $year^{-1}$ in 1994. Data collected by Gower et al. (1997) from published studies in the United States, Canada, Finland, Sweden, and China estimated average NPP at 3.6 and 1.4 Mg C ha^{-1} $year^{-1}$, for deciduous and coniferous boreal forests, respectively, values that are relatively consistent with the field measurements. Li et al. (2002), using the Carbon Budget Model of the Canadian Forest Sector (CBM–CFS2), simulated an average NPP of 1.72 Mg C ha^{-1} $year^{-1}$ for the three prairie Provinces in west-central Canada, varying from 0.72 to 2.9 Mg C ha^{-1} $year^{-1}$, depending on ecoclimatic region, forest type, age and site productivity.

Belowground NPP data are relatively rare. There are few reliable values for root productivity because the turnover rate of fine roots is difficult to measure. However, field measurements by Steele et al. (1997) at BOREAS sites (the northern study area in Manitoba and the southern study area in Saskatchewan) yielded belowground NPP estimates for mature aspen, black spruce, and jack pine stands of 0.4–0.7, 0.90–1.2, and 1.0–1.1 Mg C ha^{-1} $year^{-1}$, respectively. On the basis of published global data for the boreal forest, Gower et al. (1997) estimated an average belowground NPP of 1.1 and 1.0 Mg C ha^{-1} $year^{-1}$ for deciduous and coniferous boreal forests, respectively. Total aboveground and belowground NPP was estimated (from field measurements) at 4.7 Mg C ha^{-1} $year^{-1}$ for deciduous, broad-leaved species and 2.7 Mg C ha^{-1} $year^{-1}$ for coniferous species (Gower et al. 1997). These values are comparable to the total NPP simulated by Li et al. (2003) for western Canada, which varied between 1.5 and 3.9 Mg C ha^{-1} $year^{-1}$.

Most of the boreal forest sites investigated by various researchers are sequestering carbon (measured as net ecosystem productivity, NEP) at annual rates of up to 2.5 Mg C ha^{-1} $year^{-1}$ (Black et al. 1996; Jarvis et al. 1997; McCaughey et al. 1997; Blanken et al. 1998; Chen et al. 1999). The values obtained depend primarily on latitude, soil type, forest type, and successional stage. Not all boreal ecosystems are sequestering carbon, however. The NEP measurements over periods of up to 5 years in northern Canada in the BOREAS experiment (Sellers et al. 1997) have demonstrated that a few old-growth coniferous stands may be carbon neutral (Goulden et al. 1998) and in warm and cloudy years can be a carbon source (Lindroth et al. 1998), losing carbon at a rate of up to 1.0 Mg C ha^{-1} $year^{-1}$.

Adjacent stocks (lakes, peatlands)

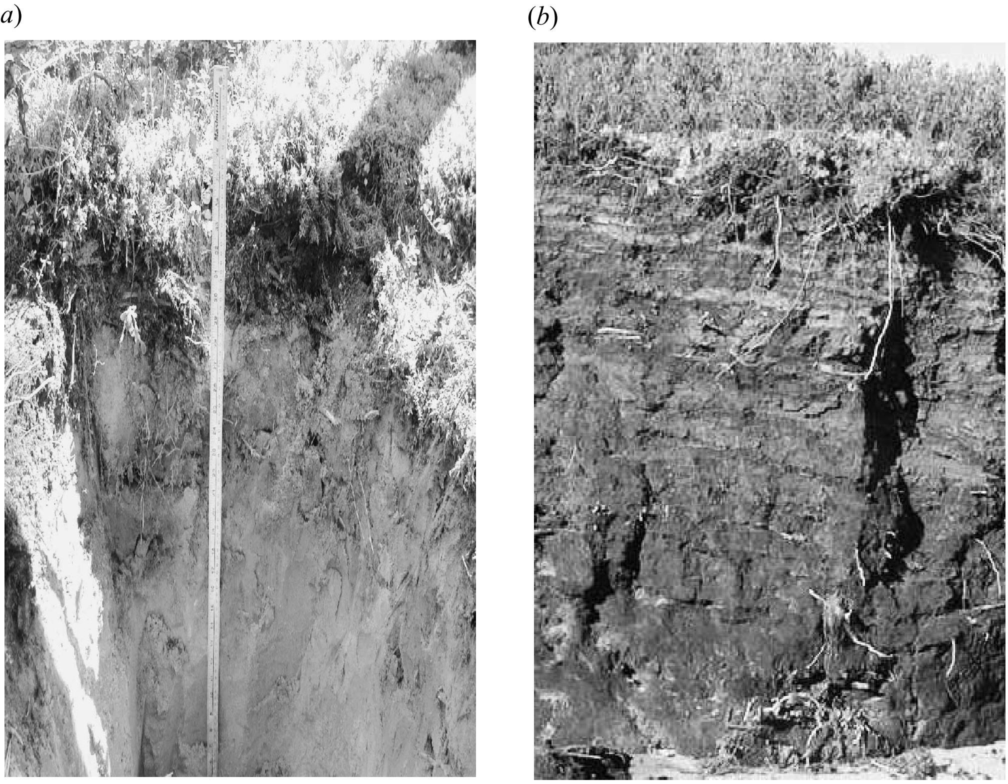
Lakes

Very few data exist for carbon storage in lake sediments in the boreal forest or elsewhere. According to Molot and Dillon (1996), 120 Pg of carbon may be stored in the sediments of boreal lakes. Campbell et al. (2000) suggested 2.3 Pg C as a first estimation of the amount of carbon stored in Alberta lakes (most of which occur in the boreal region) and estimate the average annual carbon sequestration rate to be 0.23 Tg year^{-1} (1 Tg = 10^{12} g). On the basis of data mainly from Minnesota lakes, Dean and Gorham (1998) estimated the global carbon sequestration rate in lakes and inland seas at 42 Tg year^{-1}. Since lake sediments have the potential to store a significant amount of carbon over very long periods of time, more research is needed on this component of the boreal carbon cycle.

Peatlands

Peatlands represent one of the largest terrestrial carbon reservoirs in the world. Boreal peatlands are estimated to contain 61% of the boreal carbon stocks in both Canada and around the globe (Table 20.1). Boreal and subarctic peatlands have accumulated about 400–500 Pg C during the Holocene (the last 12 000 years; e.g., Gorham 1991; Zoltai and Martikainen 1996; Clymo et al. 1998; Roulet 2000). Today, peatland carbon stocks are equivalent to about one-third of the world's soil carbon pool (1395 Pg C; Post et al. 1982) or more than double the amount of carbon stored in upland boreal forest soils (199 Pg C; Apps et al. 1993*a*; Fig. 20.7). In contrast, only about 11% of the global vegeta-

Fig. 20.7. Soil profiles: (*a*) upland forest soil profile with a very shallow organic carbon rich layer; (*b*) peatland soil profile with a very deep organic carbon rich layer. Most of the carbon in the boreal region is stored in organic peatland soils.



(*a*)

(*b*)

Photos by J.S. Bhatti

tion carbon pool (610 Pg C; Schimel 1995) is stored in the biomass of both upland and lowland areas of boreal forest (i.e., 64 Pg C; Apps et al. 1993*a*).

Across Canada, an estimated 111 Mha, or 12% of the total land base, is covered by peatlands (NWWG 1988) that store 103–184 Pg C (Ovenden 1990; Kurz et al. 1992; Apps et al. 1993*b*). In continental western Canada (Alberta, Saskatchewan, and Manitoba), peatlands cover about 20% of the land base, or 365 000 km^2, and store 48 Pg C: 42 Pg C as peat, and 6 Pg C as living aboveground biomass (Vitt et al. 2000). Estimates of peatland area in the Former Soviet Union (FSU) vary widely from 77 to 165 Mha (Botch et al. 1995), and peatland carbon stocks in the FSU are estimated at 215 Pg C (Botch et al. 1995; Kobak et al. 1998).

Global estimates of annual carbon accumulation in peatlands range from 45 to 210 Tg C $year^{-1}$ (Bramryd 1979; Armentano 1980). Regional rates of carbon accumulation range from 0.14 to 0.28 Mg C ha^{-1} $year^{-1}$ (Gorham 1991; Mäkilä 1997; Rapalee et al. 1998; Vitt et al. 2000), and may vary with changes in moisture, soil temperatures, reduction–oxidation conditions (Reader and Stewart 1972; Clymo 1984), acidity and alkalinity (Szumigalski and Bayley 1997; Thormann and Bayley 1997; Thormann et al. 1999), species composition (Malmer 1986; Johnson and Damman 1991), and (or) litter quality (Yavitt and Lang 1990; Valentine et al. 1994; Updegraff et al. 1995; Yavitt et al. 1997). Recent carbon balance studies have revealed that individual peatlands may switch from net carbon sinks to net carbon sources on an annual basis (Alm et al. 1997; Rivers et al. 1998; Waddington and Roulet 2000; Lafleur et al. 2001), perhaps responding to slight differences in weather and (or) drainage conditions.

Natural factors affecting forest carbon stocks

Disturbances and forest age

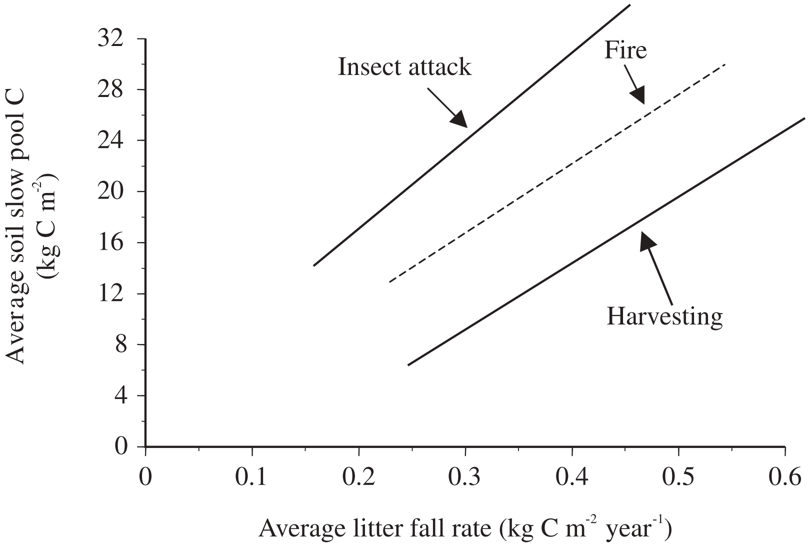
Disturbances alter forest productivity, may release carbon directly into the atmosphere (through combustion), and transfer large amounts of carbon from biomass into detritus, soils, or forest products (Kurz and Apps 1999). For example, in Canada, large forest fires from 1959 to 1999 directly released 3–115 Tg C annually (mean 27 ± 6 Tg C $year^{-1}$; Amiro et al. 2001*b*). Combustion losses of carbon range from 10 to more than 50 Tg C ha^{-1} (Kasischke et al. 2000). Intense fires may combust nearly all aboveground biomass, ground vegetation, and forest floor while leaving behind a nearly bare, ash-covered mineral soil (see Chap. 9). Fire has a substantial influence on the post-fire emissions from the soil. Schlentner and Van Cleve (1985) estimated that in mature black spruce forests approximately 20% of CO_2 emissions from soil is from decomposition and 80% is from plant root respiration. Studying post-fire carbon emissions from mature black spruce forests, Richter et al. (2000) found a threefold increase in CO_2 emissions due to a higher rate of decomposition. Post-fire release of carbon due to decomposition is estimated to be equivalent to the direct emissions during combustion in some from boreal ecosystems (Amiro et al. 2001*b*).

In continental boreal regions, peatland fires may release up to 5.5 Tg C $year^{-1}$ to the atmosphere through a combination of organic matter combustion and altered decomposition (Zoltai et al. 1998; Turetsky et al. 2002). A significant warming has been observed at many forested peatland sites after fire (Trumbore and Harden 1997). Long-term studies of permafrost following fire demonstrate decades-long permafrost melting and 5–10°C increase in soil temperature (Viereck and Dyrness 1979). Such an increase in soil temperature and permafrost melting might greatly stimulate decomposition rates. Presuming a Q_{10} of 2 (biological reactions increase about twofold per 10°C rise in temperature) and a temperature increase of 10°C, there will be a 100% increase in the rate of decomposition. These effects may last more than a decade, with a substantial increase in CO_2 emissions following fire.

Although insects do not themselves cause significant direct emissions, they affect carbon stocks by killing trees, which leads to subsequent decay of dead organic matter (Fleming 2000; Volney and Fleming 2000). During insect outbreaks, trees are often killed either directly or indirectly because of increased susceptibility to fire or disease. When the dead trees decay or are burned, they release the carbon held in the forest ecosystem back to the atmosphere. Insect disturbances also greatly increase litter fall, thereby transferring large amounts of carbon from the living biomass to the litter and soil carbon pools (Fig. 20.8).

Disturbances also drive the age-class structure of the forest, which in turn affects the forest's ability to sequester carbon. Small changes in weather patterns over periods of years to decades and longer can change the disturbance frequency, which can produce a significant shift in the age-class structure and spatial arrangement of the forest (Gardner et al. 1996). With increasing frequency of disturbance, a greater proportion of the forest is found in younger age classes. Young and immature stands in the landscape contain less carbon than mature stands, other factors being the same. A mature stand may be more susceptible to mortality from insects, diseases, and possibly forest fires, which increases inputs to detrital pools and consequently increases the release of CO_2 to the atmosphere through decomposition of the larger detrital pool.

Fig. 20.8. Simulated average C stocks in litterfall and soil (slow pool) for three disturbance types in Saskatchewan boreal forest sites (modified from Apps et al. 2000).

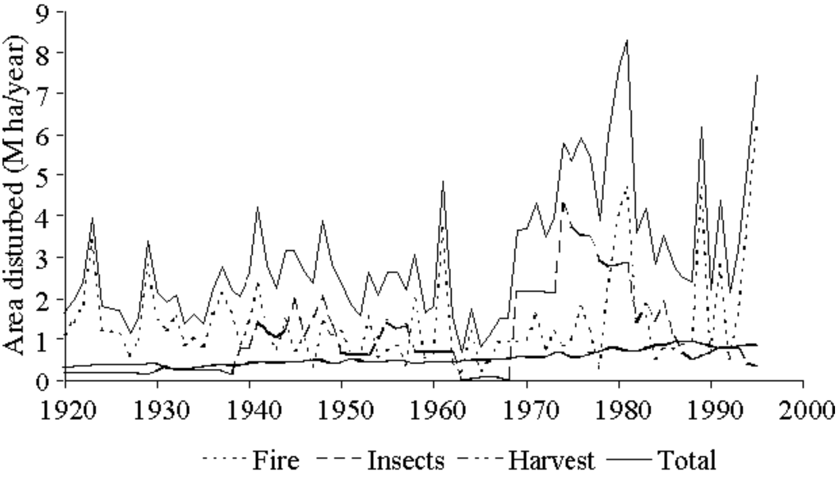


Since the late 1980s, Kurz and Apps (1999) have been developing a computer simulation model (the Carbon Budget Model of the Canadian Forest Sector) that describes the carbon dynamics of Canada's forest ecosystems. The model simulates forest growth, mortality, decomposition, the effects of natural disturbances (fires, insects, and diseases), and the impacts of direct human activities (harvesting, product processing, planting, and forest protection) on the forest carbon cycle. The effects of disturbances on the age-class structure of forest stands and carbon releases to the atmosphere and forest floor are also calculated. Their research indicates that disturbance rates and carbon sequestration have varied over the last century (Kurz et al. 1995*a*; Kurz and Apps 1999). There was a high rate of boreal forest disturbance in Canada in the late 1800s, which resulted in the establishment of large cohorts of young stands in the early 1900s. As a result, the forest was sequestering large amounts of carbon by the 1920s. From 1920 to 1970 the disturbance rates were fairly low (Fig. 20.9). By 1970 the forest was increasingly composed of mature stands of trees, which led to a general decrease in the amount of carbon being sequestered. From about 1970 to the present, many of these stands have become mature or over-mature and are carbon neutral. However, at the same time there has been a dramatic increase in disturbances, and some of these older stands are being replaced by new young stands. High combustion, decomposition, and respiration losses of carbon accompany these disturbances. As a result of changes in disturbance-driven age-class structure, Canada's total forest ecosystems were a net carbon sink from 1920 to1979 and a small net carbon source from 1980 to the present (Kurz and Apps 1999).

Other natural factors

Forest carbon stocks are also influenced directly by weather and regional climatic shifts, although in a less dramatic way than the disturbance-driven carbon fluxes. In general, forest productivity (and thus carbon accumulation) may be enhanced by warmer or

Fig. 20.9. Disturbance rates in Canada since 1920. Disturbance rates rose dramatically in 1970 after several decades of fairly low disturbance (data compiled from Kurz et al. 1995*b*, with additional data from W.A. Kurz and M.J. Apps, Canadian Forest Service, Victoria, British Columbia, personal communication, 2001).



longer growing seasons (where water and nutrients are not limiting) and diminished in cooler or shorter growing seasons (and in drier years in regions where water is limiting).

Warmer temperatures also affect the permafrost that underlies parts of the boreal forest. Temperatures increases of about 1°C across the boreal forest over the past 100 years or so (Campbell and McAndrews 1993) have resulted in widespread melting of permafrost in peatlands (Halsey et al. 1995; Vitt et al. 2000). Permafrost melt increases carbon accumulation in peatlands by increasing bryophyte primary production (Turetsky et al. 2000). To date, about 2630 km^2 of permafrost has melted in the boreal peatlands of western Canada, increasing regional carbon accumulation by approximately 0.1 Tg C $year^{-1}$ (Turetsky et al. 2002).

Potential impacts of climate change on forest carbon stocks

There is much uncertainty concerning the net impact of climate change on boreal forest carbon stocks. Some models (e.g., Smith et al. 1993) suggest that once the boreal forest achieves equilibrium with the new (and assumed stable) climate, the carbon stocks will be greater due to an increase in forest area from the migration of forest species into the presently unforested tundra and increased forest productivity. Neilson (1993), however, argues that increased drought stress would result in the boreal region becoming a long-term source of atmospheric C. Rizzo and Wiken (1992) also suggest that warmer and drier mid-continental conditions in Canada will result in greater losses of boreal regions in the south (due to grassland and agricultural encroachment) than gains in the north (due to forest migration into present tundra), which would lead to a net reduction in carbon stocks (Apps et al. 1993*a*).

In the shorter term (the next 100 years), it is likely that climate change will be accompanied by an increase in natural disturbances (fire, insects, disease, windthrow) that will reduce forest carbon stocks by releasing large quantities of carbon to the atmosphere. While regeneration of the disturbed forest will sequester some of this carbon, it is not known if the increased carbon uptake of the younger forest (the spatial structure and growth characteristics of which will be determined by the altered climatic and environmental conditions) will equal the carbon losses of the forest it replaced (Apps et al. 1993*a*). Modelling results by Bhatti et al. (2001) emphasize the importance of disturbance to carbon stocks in different types of forests (Table 20.2). Carbon stocks in aboveground biomass and litter were approximately 50% less in periods of high disturbance rates than in periods of infrequent disturbance.

The response of soil carbon stocks will also be important to the net carbon balance under a changed climate. For example, simulation results from Finland using SIMA (a gap-phase type stand dynamics model; see Chap. 14) show that total carbon reserves in forests decreased due to a decline in the carbon content of soil organic matter (SOM; Mäkipää et al. 1999). Changed climatic conditions (i.e., elevated temperature and increased precipitation) accelerated the soil surface carbon flux, resulting in a 30% decrease in soil C. This result is consistent with experimental studies, where elevated temperature greatly increased soil respiration (Peterjohn et al. 1995) and where both elevated temperature and soil moisture resulted in accelerated mineralization of SOM (Goncalves and Carlyle 1994). Many models have predicted that turnover of organic

Table 20.2. Simulated biomass, litter production, and soil C content for different stands using the simplified CBM–CFS2 model for different frequencies of random disturbance (modified from Bhatti et al. 2001).

	Coniferous (3 productivity levels)				
Stand attribute	High	Medium	Low	Mixed wood	Deciduous
Average biomass C (kg m^{-2})					
Low period	5.02	2.60	2.10	5.95	4.92
Medium period	4.99	2.57	2.08	5.82	4.79
High period	2.34	1.20	0.97	2.77	2.29
Average litter production C (kg m^{-2} year^{-1})					
Low period	0.63	0.41	0.35	0.46	0.51
Medium period	0.58	0.35	0.31	0.39	0.43
High period	0.31	0.20	0.17	0.28	0.26
Average soil C (kg m^{-2})					
Low period	30	19	17	30	24
Medium period	31	20	17	31	26
High period	28	18	15	28	23

matter in soils is accelerated by elevated temperature, which may lead to a decreased carbon stock of SOM on a global scale (Jenkinson et al. 1991; Cao and Woodward 1998).

Limited data and understanding of the influence of changing environmental conditions and disturbance (including fires and permafrost melting) on the carbon budget of peatlands over short and medium time scales (10–100 years) hinder predictions of the changes in the carbon sink–source relationships under a changing climate. The projected warming and associated changes in precipitation will influence both net primary production and decomposition in peatlands, but how global warming will directly influence peatland carbon stocks remains uncertain (Moore et al. 1998). Global change may have indirect implications for peatland carbon sinks through increased permafrost melt and fire activity. Permafrost melt tends to increase peatland carbon stocks through increased bryophyte productivity but also appears to increase heterotrophic respiration (Turetsky et al. 2000). Peatland fires result in decreased net primary production and elevated post-fire decomposition rates, but little is known about the recovery of peatland carbon balance after fire (Auclair and Thomas 1993; Dixon and Krankina 1993; Wardle et al. 1998; Zoltai et al. 1998; Turetsky and Wieder 2001).

Human impacts on forest carbon stocks

Land-use change and land-use practices

Land-use change is usually associated with a change in land cover and often results in a change in carbon stocks. Land-use changes affecting the boreal zone include forest clearance (e.g., for agriculture or roads), reforestation of land previously cleared for crops or pasture, flooding of forested areas for reservoirs, and wetland drainage. Such changes are distinguished from land-use practices which do not result in a long-term change in cover, but which may also affect carbon stocks. Land-use practices in the boreal region include forest protection (e.g., fire and insect protection, and establishment of ecological reserves) and productivity enhancement (fertilization, tree species selection, and reduction of regeneration delays after disturbance). Some of these land-use practices will be discussed below.

In Canada, the boreal forest has been subjected to increasing clearance for agriculture (mainly at the southern boundary; Fitzsimmons 2002), roads, survey lines, oil and gas wellheads, and other purposes. From 1990 to 1998, 54 000–81 000 ha year^{-1} of forest were cleared for various activities (Robinson et al. 1999). When a forest is cleared, the carbon stocks in aboveground biomass are either removed as products, released rapidly by combustion, or released slowly through microbial decomposition. Carbon stocks in the soil are also affected, although this effect depends on the subsequent treatment of the land. Clearance followed by cultivation may result in large decreases in soil carbon. For example, in boreal sites in eastern Canada, land cleared for agriculture contained 22% less soil carbon than the adjacent forested land (Carter et al. 1994).

Flooding forested and wetland areas in the boreal zone for hydroelectric reservoirs generates massive fluxes of dissolved organic carbon (DOC) into the water, accelerates peat decomposition, and increases methane and carbon dioxide fluxes to the atmosphere (Duchemin et al. 1995; Kelly et al. 1997; Schindler 1998). For example, Kelly et al. (1997) experimentally flooded a boreal wetland in Ontario, causing the carbon dynamics of the site to change from a sink of 6.6 g C m^{-2} year^{-1} to a source of 130 g C m^{-2} year^{-1}. Turetsky et al. (2002) estimated that 0.8 ± 0.2 Tg C year^{-1} is released from approximately 780 km^2 of hydroelectric reservoirs in peatlands across western boreal Canada.

When wetlands are drained for conversion to agriculture and (or) pasture, or to increase forest productivity, previously waterlogged soils become exposed to oxygen. Carbon stocks, which are resistant to decay under the anaerobic conditions prevalent in wetland soils, can then be lost by aerobic respiration (Minkkinen and Laine 1998). When drained peatland soils are farmed, as much as 10–20 Mg C ha^{-1} year^{-1} or more can be lost to the atmosphere (Armentano and Menges 1986). When drained peatlands are used for forestry, however, the reaccumulation of carbon in vegetation must also be considered, and the impact on the net carbon balance is less clear (IPCC 2001*b*).

Protecting the forest from fire and insects can temporarily preserve the carbon stocks in the forest. Protection of the entire boreal forest area, however, is not practical (Weber and Stocks 1998), nor may it be ecologically desirable, since the boreal forest is a disturbance-dependent ecosystem. When disturbances do arise, the seeding or planting of trees in the disturbed area can hasten forest regeneration and diminish the gap between the time of the disturbance and the time when the regenerating forest again recaptures carbon from the atmosphere (Kurz et al. 1995*b*). Fertilizers may also enhance growth and carbon sequestration rates, although widespread use of fertilizers is not practical or desirable from the point of view of both human and ecosystem health (Kimmins 1992). In addition to the detrimental effects indicated in Box 20.2, applications of nitrogen fertilizers to forests have large carbon emission costs associated with the production and distribution of the fertilizers themselves, and can lead to significant increases in emissions of N_2O, a potent greenhouse gas, a phenomenon well known in agricultural systems. Planting faster growing tree species such as hybrid poplar (*Populus* spp.) may also increase carbon sequestration rates on disturbed areas, but again this option is not practical on a wide scale because of economic costs and biodiversity concerns (van Kooten et al. 1993).

Forest management

Forest management activities influence the rate of carbon sequestration in a forest at both stand and landscape levels. At a stand level, silvicultural treatments, harvesting, and other actions that affect the regeneration delay and vigour of the growing trees are important (see Chap. 13). At a landscape level, management actions such as fire suppression, insect reduction programs, overall management objectives (including the rate of timber harvest) are important and are described in greater detail in Chap. 11.

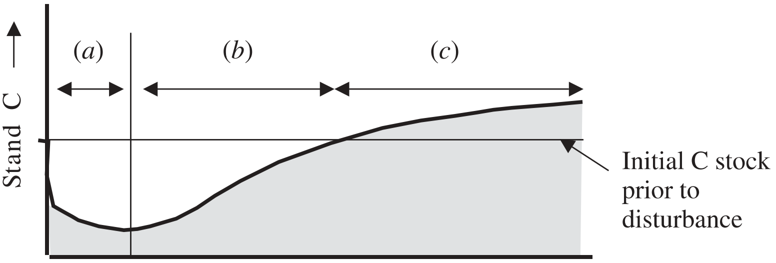
Stand-level management

Various management factors influence the carbon stores in the biomass and soil of forest stands. Stand-level actions that affect the storage of carbon include pre-harvest silvicultural techniques (site preparation, planting and spacing, thinning) as well as harvest methods (clearcutting or partial cutting, and factors that affect how much and what type of material is removed from the site). An increase in sequestration capacity is gained through an increase in carbon uptake, a decrease in carbon release, or some combination of these. After disturbance, a forest stand's net carbon balance is a function of the photosynthetic uptake less the autotrophic and heterotrophic respiration occurring. While a stand is young, the losses through decomposition outweigh the gains through photosynthesis, resulting in a net source (Fig. 20.10). Not until the rate of uptake by the vegetation overcomes this decomposition (which is affected by the type of stand-replacing disturbance) does the stand act as a sink. However, it is not until some later time (indicated by point c in Fig. 20.10) that the pre-disturbance carbon stocks in the stand are re-established — i.e., the stand has recaptured the carbon released by the disturbance event.

Increasing the carbon uptake can be accomplished through techniques that reduce the time for stand establishment (such as site preparation, planting, and weed control), or increase available nutrients for growth, or through the selection of species that are more productive for a particular area. Decreasing the losses can be accomplished through modification of harvesting practices such as engaging in lower impact harvesting (to reduce soil disturbance and damage to residual trees), increasing efficiency (and hence reducing logging residue), and managing residues to leave carbon on site (Binkley et al. 1997).

Fertilization has also been used to enhance stand productivity and can result in increased long-term carbon retention in trees and soils (Nohrstedt 2001); however, fertilization success is dependent on the site conditions, for example, on more fertile sites. For planting, species selection and stocking are important considerations. Depending on

Fig. 20.10. Stand-level carbon (C) dynamics in boreal forest after a disturbance. During the first phase after the disturbance (*a*), losses from decomposition and respiration exceed C uptake by plants — i.e., the stand is acting as a C source; (*b*) during this phase, plant C uptake exceeds the losses from decomposition and respiration (i.e., the stand is a net C sink), but the stand has not yet regained the C level prior to disturbance; (*c*) during this phase, the stand is a net sink, and C stocks exceed the pre-disturbance level.



the management objective, planting fast-growing species, such as hybrid poplar, can yield high carbon accumulation rates in early years (Schroeder and Kort 2001); however, for long-term sequestration, planting species adapted to the local climate may be more effective (Schroeder and Kort 2001).

Landscape-level management

The total effect of a management regime is not measured by the loss and gain of an individual stand, but rather by the summation of the losses and gains in ecosystem carbon over the entire landscape being managed. Carefully planned harvesting methods and silvicultural treatments may maximize the carbon storage in individual stands; however, the management of the entire landscape (including fire and insect suppression, timber and non-timber objectives) may enhance or negate the benefits of stand-level management (Kurz et al. 1998). Managing the rotation length of a stand affects its carbon dynamics (Cooper 1983). Liski et al. (2001) found that for Scots pine (*Pinus sylvestris* L.) and Norway spruce (*Picea abies* (L.) Karst.) in Finland, extending the rotation length resulted in sequestration of more carbon. At a landscape level, managing a forest for wood production can lead to greater carbon storage than that of natural forests (Price et al. 1997; Kurz et al. 1998) in areas where management reduces the frequency of natural disturbances.

Fire- and insect-protection activities have a strong impact on the carbon sequestration of the forested landscape. Reducing the areas affected by fire and insect mortality extends the rotation age, effectively holding carbon longer in older age classes and allowing younger, more actively sequestering stands that would have otherwise been killed to continue to sequester carbon.

The overall carbon budget of the landscape is a result of the combination of uptake and release from the various stands at the different stages of development. Not only do current and future conditions play an important role in this balance, but the potential for a landscape to sequester carbon both now and in the future also depends strongly on past conditions. At a stand level, fire, insects, and harvesting each have different effects on the residual carbon pools remaining on site and the subsequent releases of carbon through decomposition. As such, similar stands with different disturbance histories can have different amounts of carbon accumulated in the dead organic matter pools. At a landscape level, changing from a lower disturbance regime to a higher disturbance regime will result in a net carbon source, whereas migrating toward a lower disturbance regime will result in a net sink.

Different management options being examined in order to address various environmental concerns, while meeting society's need for wood products, also have implications for forest carbon. Management regimes such as the triad approach (see Chap. 12) include areas of high management intensity (e.g., plantations which are intensively managed with high-yield, shorter rotations), extensive forest management (larger areas with more traditional forest management), and protected areas (where no harvesting is performed).

Adjacent carbon pools (lakes, peatlands)

Lakes

Few data exist concerning the effect of harvesting on the net carbon budget of lakes because, until recently, most research on aquatic impacts has focused on streams rather than lakes. The impact of harvesting on the levels of DOC and total suspended solids in lakes varies, depending on the size of the clearcut, the drainage ratio of the lake (i.e., the size of the lake compared with the size of the catchment), and the morphology of the watershed (Carignan and Steedman 2000). As described more fully in Chap. 10, clearcutting can lead to hydrological flushing of DOC from the soil into surface waters and result in increased lake DOC levels (Carignan and Steedman 2000; France et al. 2000). Carignan and Steedman (2000) found this to be especially true for lakes with large drainage ratios. In a study of 116 Boreal Shield lakes in Ontario and Quebec, France et al. (2000) found that lakes in logged watersheds contained 2 mg L^{-1} more DOC than lakes in undisturbed watersheds. In their study of 800 boreal lakes (in Ontario and Quebec), Carignan et al. (2000) also found higher DOC levels in lakes with logged watersheds than in lakes with undisturbed catchments. In contrast, Prepas et al. (2001) found no evidence of increased DOC concentrations in lakes on the Boreal Plain following harvest. The net effect of logging on the total carbon budget of boreal lakes is unknown, however, and will depend on total carbon inputs (i.e., particulate matter in addition to DOC) and how much carbon is ultimately stored in the sediments, exported in lake outflow, and evaded from the water column to the atmosphere (Gennings et al. 2001). While lake carbon budgets are sensitive to catchment disturbances, such as logging and fire, they are thought to be influenced to a greater degree by climate, hydrological parameters, and lake acidity. The only published evidence of increased DOC after fire is associated with fire in the bog-laden watersheds underlain by permafrost at the northern extent of the Boreal Plain (see McEachern et al. 2000).

Peatlands

Peatland drainage and peat extraction for horticultural products has increased over the past few decades (Waddington and Price 2000). In 1997, 234 Gg (1 Gg = 10^9 g) of organic matter was extracted from peatlands across western boreal Canada (Statistics Canada 1997). This material subsequently is oxidized under aerobic conditions when used as a potting medium and soil amendment. Harvested peatlands also have higher rates of CO_2 emission than undisturbed peatlands (Waddington and Price 2000). While restoration appears to enhance carbon accumulation relative to harvested sites, it cannot restore peatland carbon sinks to their pre-disturbed state for several decades or even centuries (Waddington and Price 2000). Other human activities, such as oil sand development in northern Alberta, also influence peatland carbon sinks by altering plant production and microbial respiration or by exporting peat from its natural waterlogged setting. Contemporary carbon budget assessment in western continental peatlands suggests that natural disturbances (i.e., fire and permafrost melt) are more important to peatland carbon sinks than direct human activities, such as harvesting and oil sands development (Turetsky et al. 2002). However, the majority of peat extraction in North America occurs in New Brunswick and Quebec, and may be a more significant factor in peatland carbon balance in eastern Canada.

Wood products

Bioenergy

Biomass energy can be used to reduce greenhouse gas emissions from fossil fuels by providing an alternative renewable source of energy. Efficiently produced bioenergy from sustainably managed forests is thought to be almost carbon neutral (Mercier 1998) because the carbon stored in the biomass (and released during combustion) was originally sequestered from the atmosphere and will be replaced as the forest regenerates (Schlamadinger et al. 1997). Fossil fuels, on the other hand, are non-renewable and result in constant accumulation of carbon in the atmosphere.

In Canada, bioenergy is becoming a significant source of renewable energy (ahead of wind and solar energy) and provides approximately 6% of the total primary energy supply and 7% of primary residential heating (Mercier 1998; Hauer et al. 2001). The main producer and consumer of bioenergy in Canada is the forest sector, particularly the pulp and paper industry. The use of bioenergy in the pulp and paper industry has steadily increased in recent decades, while the use of fossil fuels has declined in the past 20 years (Apps et al. 1999) (Fig. 20.11). The pulp and paper industry is expected to double its use of bioenergy sources between 1990 and 2020 and significantly reduce its use of fossil fuels (NRCan 1998; Hauer et al. 2001). Although the forest sector currently self-generates about half of the power that it uses in some regions (Gardner 1999; see Chap. 19), it is constrained in many cases from achieving economies of size in power generation by either an inability to sell excess power into the provincial grid or a lack of fibre (Curtis 2000) to sustain the supply.

As shown in Table 20.3, energy from wood residues can compete with fossil fuels and purchased electricity. This conclusion needs careful scrutiny, however. First, wood residue prices are based on average, not marginal, costs and are available only for small-scale operations where wood is easy to obtain. At a larger scale, much higher raw material (wood) costs can be expected. Second, wood fibre prices vary significantly by

Fig. 20.11. Energy-use efficiency of the pulp and paper industry in Canada (10-year average for each decade). Energy from biomass carbon has steadily increased while energy from fossil fuel carbon has declined since the 1970s (modified from Apps et al. 1999).

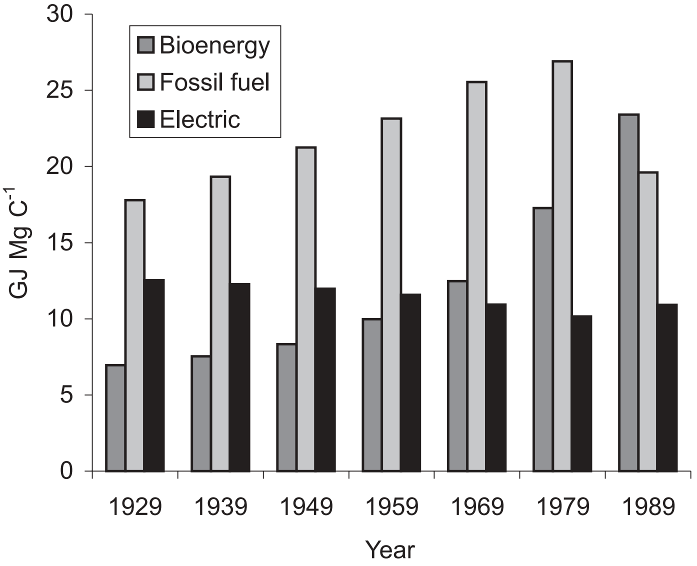


Table 20.3. Energy price comparisons for British Columbia, in 1997 $ (based on FST 1999 and authors' calculations).

Wood residues (assumed conversion factor = 18 GJ per dry Mg)	$ GJ^{-1}
Wood residue from pulp and paper mills	1.00
Wood residue from wood industry	0.56
Wood waste plus plantation wood	2.82
Fossil fuels (based on natural gas, boiler efficiency of 85%, C emission factor of 0.050 Mg GJ^{-1})	
Fuel price	1.73
Electricity (at $0.039 per kW h)	10.84

region, depending on residue surpluses or shortages, and environmental regulations. Regional values are not currently available for comparison (Gardner 1999). If fast-growing plantations are included, estimated costs are $2.82/GJ, which is more expensive than fossil fuels but still cheaper than purchased electricity, which (in boreal regions) is usually generated from hydroelectric sources.

In Canada, the high capital cost of infrastructure, regulation of the electricity market, and the relatively low cost of fossil fuels restrict the economic viability of substituting biomass for fossil fuels in power generation. When global climate change impacts, future energy requirements, availability of supply, and social and environmental values are considered, it is found that the benefits of renewable energy sources such as wood biomass outweigh the costs in some, but not all situations.

Fast and slow turnover forest products

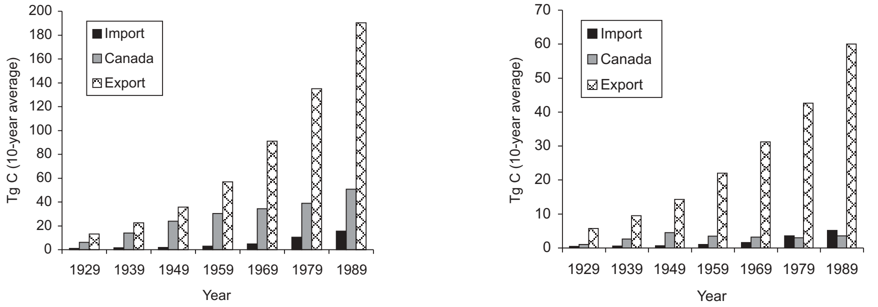
Although all wood products release carbon back to the atmosphere through oxidation, pools that delay this release act as temporary storage pools, reducing the proportion of carbon leaving the terrestrial system on an annual basis (IGBP 1998). For a given production rate, the amount of carbon stored in wood products is primarily determined by the life span of these products. Thus slow-turnover pools store carbon for longer periods than fast-turnover pools. Slow-turnover pools include lumber and other long-term structural building products; fast-turnover pools include pulp and paper products, fuelwood, and scrap lumber. Discarded wood products and waste products may eventually end up in landfills that also have a slow turnover rate. To delay the emissions of carbon back to atmosphere from shorter life-span pools, the product's turnover time may be extended or the end fate of the product changed. Increased recycling of forest products (see Chap. 19) effectively acts to increase the residence of carbon in these products, especially products with a shorter life span.

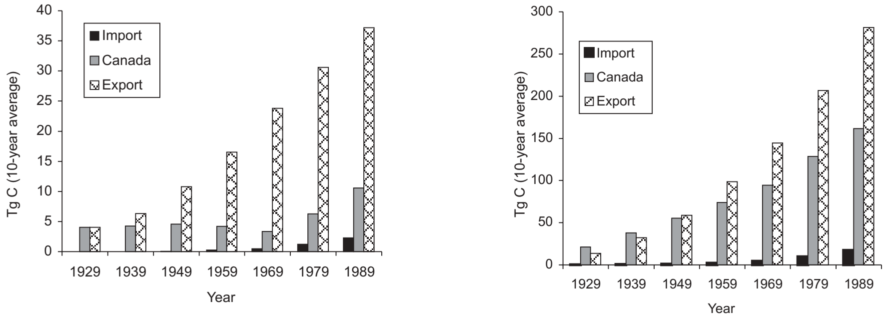
Apps et al. (1999) performed a 70-year simulation analysis of the carbon dynamics of the Canadian forest products sector for the years 1920–1989. Their results suggest that the total carbon in forest products and landfills reached 836 Tg C by 1989 (Fig. 20.12). Pools with a long turnover time (slow pool and landfill) are, not surprisingly, the largest storage pools, containing 86% of the total.

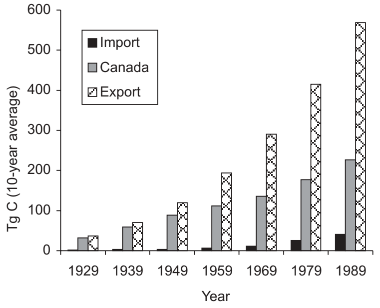
Over the 70-year analysis period, a total of 1940 Tg C was harvested from Canadian forest ecosystems, and an additional 52 Tg C was imported as products. In 1989, an estimated 837 Tg C, or 43% of the cumulative harvested biomass carbon, was retained in forest product pools and landfills in Canada and abroad. Landfills contained 461 Tg C or 55% of the retained carbon (Apps et al. 1999). These simulations suggest that landfills may provide important long-term storage for carbon, especially under anaerobic conditions. Price et al. (1997) similarly found that landfill repositories of residues played an important role in the net carbon budget of an operational forest enterprise. Micales and Skog (1997) studied the decomposition of forest products in landfills and reported that United States landfills may serve as large carbon sinks, effectively preventing large quantities of carbon from being released back into the atmosphere.

The carbon stocks in the entire Canadian forest sector (excluding peat deposits) in 1989 is estimated to be 86.6 Pg C, of which 71.3 Pg C is found in the dead organic matter of litter and soils, 14.5 Pg in living biomass, and only 0.8 Pg C in forest product stocks (FPS). Although the FPS carbon stocks contain less than 1% of the total forest sector carbon, they grew by nearly 25 Tg C year^{-1} in 1989 (Apps et al. 1999). The FPS carbon stocks thus play a significant role in the net forest sector exchange with the atmosphere and offset more than one-third of the net carbon release from Canada's forest ecosystems.

Fig. 20.12. Carbon stocks in wood product pools in the Canadian forest sector between 1920 and 1989. (Data modified from Apps et al. 1999.)


(*a*) Slow pool (*b*) Fast pool


(*c*) Pulp and paper pool (*d*) Landfill pool


(*e*) Total pools

Economic considerations

Policy background

Concern about anthropogenic emissions of GHGs, particularly CO_2, led the World Meteorological Organization and the United Nations Environment Program to jointly establish the Intergovernmental Panel on Climate Change (IPCC) in 1988. The first IPCC report was published in 1990, and was instrumental in the subsequent signing of the United Nations' Framework Convention on Climate Change (FCCC) at the Earth Summit in Rio de Janeiro in June 1992 (see Chap. 1). The Convention committed signatories to stabilize atmospheric CO_2, with developed countries to reduce emissions to their 1990 levels by 2000 — a goal that was not achieved.

It was recognized as early as 1995, at the first Conference of the Parties (COP1) to the FCCC in Berlin, that the 1992 Rio de Janeiro commitments might not be enough to mitigate global warming. The IPCC's second assessment report in 1996 (IPCC 1996), which was endorsed by COP2, created an impetus for taking more drastic action. Therefore, at COP3 at Kyoto on 11 December 1997, industrialized countries agreed to reduce CO_2 emissions by an average of 5.2% from the 1990 level by 2008–2012, which became known as the commitment period.[3] The Kyoto Protocol (Box 20.3) is seen as a first, but necessary, step towards international commitment to dealing with the threat of global climate change. There have been five other COPs since Kyoto (see discussion below), and the IPCC's third assessment report was released in 2001 (IPCC 2001*c*).

In order for the Kyoto Protocol to go into effect, 55 countries must ratify the Protocol, and of industrial countries, those that ratify must account for 55% of total Annex B countries' 1990 CO_2 emissions. Poland, Canada, and New Zealand were some of the more recent countries to ratify Kyoto as of 24 February 2003, thereby bringing the number of countries that have ratified to 105 and the developed countries' proportion of 1990 emissions to 43.9%. The pressure is now on Russia to ratify the Protocol, since it accounts for 17.4% of 1990 CO_2 emissions, and with the United States and Australia having decided not to ratify, Russian ratification is required before the Kyoto Protocol can come into force.

Canada has agreed to reduce CO_2 emissions by 6% from the 1990 level by 2008–2012. Canadian GHG emissions by sector, expressed as CO_2 equivalents, are provided in Fig. 20.13. In 1990, Canada generated 607 Tg of CO_2-equivalent emissions, implying that Canada must reduce emissions to 571 Tg by 2008–2012.[4] By 2000, after a period of economic expansion, Canada's GHG emissions had increased by 19.6%, to 706 Tg (Olsen et al. 2002). Business-as-usual (BAU) emissions are projected to reach 802 Tg annually by 2010, approximately 40% above Canada's Kyoto commitment (Fig. 20.13).[5] Clearly, considerable effort will be required, within a very short time frame, to achieve the emissions-reduction objective. How this is to be done is explained in the Government of Canada's (2002) implementation plan (discussed further below).

In October 2000, the government released the National Implementation Strategy on Climate Change and the First National Climate Change Business Plan to address Canada's strategy for implementing the Kyoto Protocol. Under Action Plan 2000, which was announced at the same time, \$500 million was set aside to provide subsidies for programs and measures for reducing CO_2 emissions. The projected reductions were expected to account for more than one-fifth of the Kyoto gap, or some 50 Tg CO_2 annu-

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[3] Industrialized countries that signed the Kyoto Protocol included those in Annex I of the FCCC plus countries of the ex-Soviet bloc. These countries are listed in Annex B of the Kyoto Protocol.

[4] In this section, the units are expressed in terms of CO_2. Divide the units of CO_2 by 3.7 to obtain the approximately equivalent weight in terms of C.

[5] These projections are based on earlier government reports and are sensitive to assumptions about population growth (including rates of immigration), economic growth, changes in energy prices, the rate of adoption of energy-saving technologies, and other factors (see NRCan 1999*b*; Goncalves 2001). Current projections indicate that the gap between the Kyoto requirement and BAU emissions is 240 Tg rather than the 231 Tg indicated here and in Fig. 20.13.

Box 20.3. The Kyoto Protocol

The Kyoto Protocol establishes legally binding targets and timetables for reducing greenhouse gas (GHG) emissions in developed countries. The GHG reduction commitments for the first commitment period (2008–2012) for key boreal signatories of the protocol are as follows (expressed as a percentage of 1990 emission levels): Canada 94%, Norway 101%, Sweden 92%, Finland 92%, Russian Federation 100%.

The Protocol recognizes that land use, land-use change, and forestry affect the net GHG balance (sources and sinks). Articles 3.3 and 3.4 provide forest-related mechanisms for compliance with the above targets:

- Article 3.3 requires an accounting of C stock changes resulting from land-use change (Afforestation, Reforestation, and Deforestation, ARD) since 1990;
- Article 3.4 allows countries to include, if they choose to do so, carbon gains/losses due to forest management practices since 1990;
- Each Party must stipulate by 2006 whether or not they will include forest management practices in their reporting; and
- If included, sinks due to forest management can be used to offset any net ARD source up to a cap of 9 Tg C year^{-1}. Any remaining sink is then credited up to a country-specific cap (Canada's cap is 12 Tg C year^{-1}).

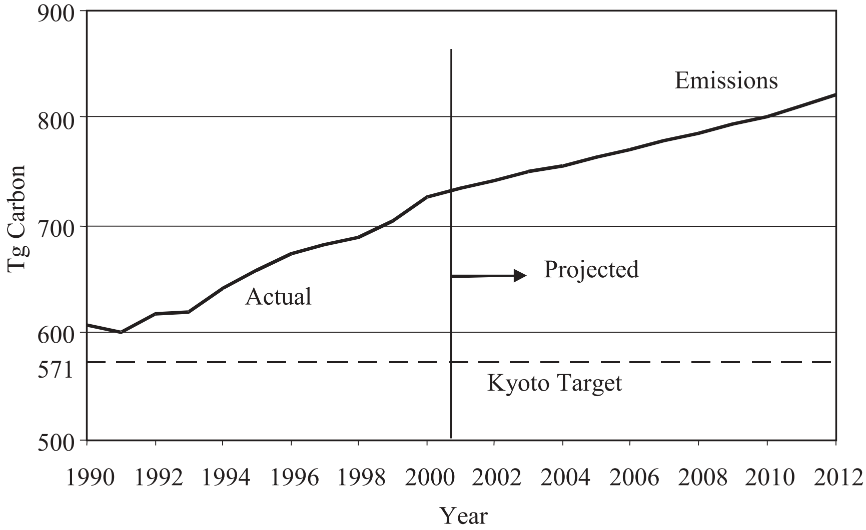
Definitions:

- *Afforestation* — the direct human-induced conversion of land that has not been forested for a period of at least 50 years to forested land through planting, seeding, and (or) the human-induced promotion of natural seed sources.
- *Reforestation* — the direct human-induced conversion of non-forested land to forested land through planting, seeding and (or) human-induced promotion of natural seed sources, on land that was forested but that has been converted to non-forest land. For the first commitment period, reforestation activities will be limited to reforestation occurring on those lands that did not contain forest on 31 December 1989.
- *Deforestation* — the direct human-induced conversion of forested land to non-forested land.
- *Forest management* — a system of practices for stewardship and use of forest land aimed at fulfilling relevant ecological (including biological diversity), economic, and social functions of the forest in a sustainable manner.

Additional information on the Kyoto Protocol including recent developments and negotiations can be found on the United Nations Framework Convention on Climate Change website, http://unfccc.int/.

ally (Goncalves 2001, p. 55; as modified, Goncalves 2002, p. 12). Subsidies and voluntary initiatives feature prominently in achieving this amount of emissions reduction. To this are added an additional 30 Tg CO_2 annually to be sequestered in terrestrial sinks (discussed below), implying that 80 Tg CO_2 per year (one-third of the Kyoto target) has already been accounted for by actions implemented in Action Plan 2000 and Budget 2001. Of the remaining 160 Tg required annual reduction in CO_2 emissions, the Government's October 2002 implementation plan (Goncalves 2002) calls for some

Fig. 20.13. Actual, projected, and Kyoto target emissions, 1990–2012, Canada.



90–100 Tg to be achieved through a variety of regulations, as well as program subsidies where reductions are achieved through modification of behaviour.

Emissions trading

Both domestic and international emissions trading are part of Canada’s implementation plan (Goncalves 2002, pp. 28–32, 42–44). The proposed domestic emissions trading system will target 55 Tg of the gap, with emission reduction targets established through negotiated covenants with a regulatory or financial backstop, although details are still under development. It is expected that offsets will be part of the emissions trading system, including forestry and agricultural sinks. In order to establish a market for CO_2 emission permits, the government needs to establish property rights to carbon, set the total amount of emission permits allowed (the “cap”), monitor and enforce the quota system, and encourage markets by developing clear, simple, and stable rules and procedures for conducting trades.

While it is relatively straight forward to establish a market for trading CO_2 emission permits, integrating carbon offsets or credits from terrestrial activities into such a scheme may be more problematic (van Kooten 2002). For example, two caps may need to be established, one for CO_2 emissions and another for carbon offsets, and these cannot be determined independently. Further, it may be necessary to establish an exchange rate between permanent reductions in atmospheric CO_2 (emission reductions) and temporary ones (terrestrial sinks). One approach to the latter problem, which has been used in Australia, is to permit landowners to sell carbon sink credits, but with the proviso that they are liable for future release of carbon, having to purchase CO_2 emission permits at the time the carbon is released. Emission trading schemes, with or without carbon offsets, can be national or international in scope. If the Kyoto Protocol goes into force, the international emissions trading regime will be an important means for reducing overall global emissions in an efficient manner. Countries will no doubt work to ensure that domestic markets are linked to the international market. Further details on carbon trading are provided by Henri (2001), Obersteiner et al. (2001), and van Kooten (2003).

In Canada and other countries, carbon trading outside the Kyoto framework has already started. Some large emitters of CO_2, primarily regulated utility companies that can usually pass such costs onto customers and companies that wish to purchase goodwill, have made some exploratory purchases of carbon credits that accrue as the result of forestry projects (Henri 2001). These companies are offsetting emissions in anticipation of future requirements to do so. Effective monitoring is critical to ensure that the contracted carbon credits actually materialize, but programs and activities that certify those providing the credits can reduce both the risk and the overall cost.

There will likely be a mix of initiatives by the private and public sectors to certify carbon credits. Sellers might be certified under ISO 9002 (quality systems, including forest inventory and mapping systems) and ISO 14001 (environmental management systems). They might also be certified at the national level or through non-governmental organizations (NGOs), which would certify independent companies who have the expertise to conduct carbon audits and (or) to certify and audit sellers. See Chap. 21 for a broader discussion of forest certification initiatives and how these might include carbon auditing. As the design of Canada's domestic emissions and offset trading system are developed, the details regarding the processes and organizations that will register and verify carbon credits will need to be elaborated.

A market for trading carbon sequestration credits will be an important development for several reasons. First, the futures market for carbon credits — a futures market because it deals with carbon uptake in the Kyoto commitment period (2008–2012) — establishes a price for carbon and provides a useful indicator for both the private sector and policymakers. Furthermore, such a market can be integrated into a larger system of carbon emissions trading: instead of purchasing CO_2-emission permits, companies (or countries) can purchase carbon uptake credits.

Carbon credits and the role of forestry

Under the Kyoto Protocol, a country can obtain carbon credits at home by planting trees where none grew previously, increasing the rate at which carbon is sequestered through forest management, or pursuing other land-use activities that increase carbon uptake in soils. Forests are important because they can sequester more carbon over a longer time period than agricultural sinks, with further benefits possible if account is taken of the fate of wood products (e.g., construction lumber, and paper in landfills) or wood biomass used to produce energy in place of fossil fuels. Forests store carbon by photosynthesis, thus removing CO_2 from the atmosphere as they grow, with each cubic metre of wood containing approximately 200 kg of carbon. Countries with significant forest cover may be interested in potential carbon credits related to forest management and those with large tracts of marginal agricultural land may be interested in afforestation as a means for achieving some of their agreed-upon reduction in CO_2-emissions. Canada falls into both these categories. Therefore, the remaining focus of this section is on the role of terrestrial carbon sinks and how forest management may provide national (and also market-traded) debits and credits under the Kyoto Protocol.

Under the Protocol, afforestation and reforestation occurring since 1990 may result in carbon credits, while deforestation (human-induced conversion of forestland to non-forest use) results in a carbon debit. The terms afforestation and reforestation had been the subject of debate, but as a result of $\mathrm{COP6}_{bis}$ (Bonn) and COP7 (Marrakech) negotiations, afforestation is now defined, for the purposes of the Kyoto Protocol, as human activities that encourage growing trees on land that has not been forested in the past 50 years, while reforestation refers to human activities that encourage growing trees on land that was forested but had been converted to non-forest use (see also IPCC 2000). An indication of the potential role of these activities on a global scale is provided in Table 20.4. Potentially, these activities could sequester an average of 1000–1580 Tg C year^{-1} by 2050. In addition, under Article 3.4 of the Kyoto Protocol, countries have the choice of whether to include carbon sequestered as a result of forest management (e.g., fertilizing, and fire control) toward emissions reductions, although no account is taken of carbon stored in wood products. Even before the United States withdrew from the Kyoto negotiations, economists estimated that carbon credits would need to be created for \$10–20 per Mg C to be competitive (Sandor and Skees 1999). There remains a question as to whether, even if all of the carbon fluxes (such as wood products storage) are appropriately taken into account, "additional" forest management will be a cost-effective and competitive means for sequestering carbon (Caspersen et al. 2000). As a result of $\mathrm{COP6}_{bis}$ and COP7 negotiations, however, countries can claim carbon credits from business-as-usual forest management that are likely not "additional" (i.e., do not reduce atmospheric CO_2 over and above what would have happened anyway). The role of terrestrial sinks and whether market mechanisms should treat carbon offsets the same way as emissions reduction was a source of dispute at COP6 in The Hague, Netherlands, in

Table 20.4. Global estimates of the costs and potential carbon that can be removed from the atmosphere and stored by enhanced forest management from 1995 to 2050 (data compiled from IPCC 1996, pp. 785–791).

Region	Practice	C removed and stored (Pg)	Estimated costs (\$CAN × 10^9)[c]
Boreal	Forestation[a]	2.4	26
Temperate	Forestation[a]	11.8	92
	Agroforestry	0.7	5
Tropical	Forestation[a]	16.4	149
	Agroforestry	6.3	42
	Regeneration[b]	11.5–28.7	68–152
	Slowing-deforestation[b]	10.8–20.8	
Total		60–87	

[a] Refers primarily to reforestation.

[b] Includes an additional 25% of aboveground C to account for C in roots, litter, and soil (range based on uncertainty in estimates of biomass density).

[c] Based on the conversion: \$1 CAN = \$0.65 U.S.

November 2000. The European Union countries took the view that there should be limits to the role of land use, land-use change, and forestry (LULUCF) projects so that countries would be forced to address domestic emissions reduction. The argument put forward was the fear that LULUCF projects are ephemeral and do not help to reduce the long-term, upward trend in CO_2 emissions. The opposite view is that reduction of CO_2 emissions and enhancement of carbon sinks are no different in their impact and should be treated the same on efficiency grounds (Chomitz 2000). Subsequent to the United States' withdrawal from the Kyoto process, the objections to inclusion of LULUCF projects softened (at $COP6_{bis}$ and COP7), opening the door for a much greater role for these projects.

The negotiations on forest sinks did produce caps, however. The Marrakech Accords (negotiated at COP7) give each country the choice of whether it wants to include forest management in its accounting in the first commitment period, with the decision to be made by 2006. A country that has a net debit from Article 3.3 (i.e., due to afforestation, reforestation, and deforestation (ARD)) can use credits from its forest management to offset this debit, up to 9 Tg C $year^{-1}$, provided it has chosen to include forest management in its accounting. Use of this offset is subject to the proviso that its managed forest since 1990 is a net sink at least equal to the ARD debit. The agreement allows further credit for forest management in the first commitment period up to a country-by-country negotiated cap, which for Canada is 12 Tg C $year^{-1}$ (44 Tg CO_2). The Russian Federation can claim up to 33 Tg C, Japan 13 Tg C, Germany 1.24 Tg C, Ukraine 1.11 Tg C, and remaining countries less than 1.0 Tg C. Canada currently estimates it will be able to claim approximately 5.4 Tg C annually of forest sinks (i.e, the net of ARD and forest management), and expects an estimated 2.7 Tg C per year from agricultural sinks (Goncalves 2002, p. 39). Countries, however, can only claim credit for actual carbon stock changes in the commitment period that will need to be accounted for using detailed procedures for verification purposes.

Carbon credits can also be obtained for activities in developing countries and economies in transition. Kyoto Protocol's Clean Development Mechanism (CDM) enables industrialized countries to purchase certified offsets from developing countries (that have ratified the Kyoto Protocol) by sponsoring projects that reduce CO_2 emissions or increase CO_2 removals beyond business-as-usual levels in those countries. Afforestation and reforestation are the only land-use activities that can be included under the CDM, and these contributions are capped at 1% of an Annex B country's baseline emissions.

Likewise, emission reduction units can be produced through Joint Implementation (JI) projects in countries whose economies are in transition. Any land management activity accountable under Articles 3.3 or 3.4 of the Kyoto Protocol is eligible under the JI provision. Russia and Ukraine are two countries where forest activities generating carbon credits might be undertaken as JI projects by other industrial countries.

As noted, the idea of relying on terrestrial sinks to meet international emissions reduction obligations remains controversial. Some argue that, in principle, a country should get credit only for sequestration above and beyond what occurs in the absence of C-uptake incentives, a condition known as "additionality" (Chomitz 2000). Thus, for example, if it can be demonstrated that a forest would be harvested and converted to another land use in the absence of specific policy (say, subsidies) to prevent this from happening, the additionality condition would be met. Additional carbon sequestered as the net result of incremental forest management activities (e.g., juvenile spacing, commercial thinning, fire control, or fertilization) would be eligible for carbon credits, but only if the activities would not otherwise have been undertaken (e.g., for timber supply purposes or to provide higher returns). Similarly, afforestation projects would be additional if they would not have been undertaken in the absence of economic incentives, such as subsidy payments or an ability to sell carbon credits (Chomitz 2000). The premise of additionality does not apply for Articles 3.3 and 3.4, although it does for CDM projects. Because of the controversy and difficulties in separating direct human-induced effects from natural effects, countries agreed on the use of caps in the first commitment period as an interim ad hoc solution. Considerable scientific work is ongoing to better address these issues for the next commitment period.

Costs of creating carbon credits in Canada

Standard financial calculations suggest that afforestation and reforestation in Canada would generally not yield carbon credits at a sufficiently low cost to be competitive with other investments, because northern forests tend to be marginal in terms of productivity (van Kooten et al. 1993). The reason is that such forests generally regenerate naturally (even on marginal agricultural land), making artificial regeneration redundant and (or) lowering the benefits to reforestation programs (van Kooten and Bulte 2000, p. 398). Studies indicate that only short-rotation, hybrid poplar plantations are likely to be competitive with other methods of removing CO_2 from the atmosphere (van Kooten et al. 1999, 2000).

For example, the TECAB model (Krcmar and van Kooten 2001; Krcmar et al. 2001) was used to examine the costs of carbon uptake in the grain belt – boreal forest transition zone of British Columbia. The model consists of tree-growth, agricultural activities, and land-allocation components. Carbon fluxes associated with many forest management activities (but not control of fire, pests, and disease) are included, and account is taken of what happens to the wood after harvest, including decay (see Table 20.5 for data on decay of forest ecosystem components). The study results indicate that upward of 1.5 Tg of discounted carbon (discounted at 4%) can be sequestered in the region at a cost of about $40 per Mg or less. This is a combination of planting hybrid poplar, as well as forest management activities, with wood products storage also accounted for. This amounts to an average of about 1.3 Mg ha^{-1}, or about 52 kg ha^{-1} per year of additional carbon uptake. While in general plantation forests are considered a cost-effective means of sequestering carbon (Sedjo et al. 1995; Adams et al. 1999), these results are a further indication that boreal forests are globally marginal, and silvicultural investments in them generally do not pay, even when carbon uptake is included as a benefit (van Kooten et al. 1993; Wilson et al. 1999).

While hybrid poplar planting in general appears to be the most economic option for creating carbon credits in northern climates, there remains a great deal of uncertainty about planting hybrid poplar on a large scale in Canada, mainly because it has not been done previously. Certainly there are a number of considerations related to the use of hybrid poplar plantations as part of a carbon uptake strategy, which include:

(1) Relative to native species, hybrid poplar plantations may have negative environmental impacts related to reduced biodiversity and susceptibility to disease (Callan 1998);
(2) If there are transaction costs associated with afforestation, this will increase C-uptake costs over and above what is generally estimated (van Kooten 2000);
(3) Uncertainty about (current and future) timber and fibre values and prices of agricultural products may make landowners reluctant to convert agricultural land to forestry;
(4) The potential of wood from afforested land as a biomass fuel is uncertain;
(5) Recent research suggests that planting trees where none existed previously decreases surface albedo, which offsets the negative forcing expected from carbon uptake (Betts 2000), although such planting must be on a large (perhaps unrealistic) scale before there is a noticeable albedo effect;
(6) There is a potential problem of leakage, since large-scale afforestation and (or) other forest plantations will tend to lower prices for wood fibre, with current wood lot owners (for example, in the southern United States) likely reducing their forest holdings by converting land back to agriculture in anticipation of the drop in price. These are generally ignored in calculating the costs of individual afforestation or reforestation projects. Yet, such leakages can be substantial, even as high as one-half of the carbon sequestered by the new plantations (Sohngen and Sedjo 1999).

In general, tree planting programs to generate carbon credits in Canada may be much more expensive than originally anticipated (van Kooten 2003).

While creating incentives to manage carbon stocks and carbon fluxes in forests efficiently would have positive benefits, it is unlikely that forest management in Canada will be a cost-effective and competitive means for sequestering carbon on a sufficient scale to dramatically affect Canada's emissions reduction target for Kyoto in the first commitment period. Compared with economic viability in other regions, Canada's forests are marginal. The only land-use change on forestlands that might be a competitive alternative to other methods of removing CO_2 from the atmosphere is if short-rotation, hybrid poplar plantations replace logged or otherwise denuded forests. However, the potential negative environmental effects of hybrid poplar plantations, such as their limited longevity, reduced biodiversity, disease, insect problems, and loss of scenic amenities, would need to be fully considered in assessing their suitability as an acceptable large-scale tool for carbon removals to help meeting emission reduction targets.

While there are numerous actions that forest management agencies and the forest sector can take to reduce emissions, and increase removals of carbon (see Box 20.4), enhancing carbon sequestration through intensive forest management or plantations is not likely to be an economic alternative on a wide scale in boreal ecosystems. However, given the significant role of disturbances in the boreal forest ecosystem, it is clear that the forest managers will influence carbon stock changes by their management actions, as discussed in earlier sections in this chapter.

Box 20.4. Options for minimizing carbon emissions in boreal forestry.

Boreal ecosystems may only be able to make limited contributions to carbon sequestration, owing to low biological productivity compared to other parts of the world. But their vast area and large carbon stocks mean that boreal forests will always play a major role in the global carbon budget. Government agencies, forest products companies, and forest managers have a number of tools at their disposal to minimize the emission of CO_2 and other GHGs to the atmosphere:

- minimize and optimize the use of fossil fuels:
- use fuel-efficient machinery, trucks, etc.;
- minimize transportation requirements through centralized positioning of processing facilities, and harvesting forest in contiguous rather than dispersed patterns;
- substitute fossil fuels with biofuels wherever possible;
- inventory and monitor carbon stocks in biomass, soils, peatlands, and lakes;
- promote recycling and reuse of wood and paper products;
- treat wood products with preservatives that extend their useful life;
- protect peatlands from drainage, fire, and land-use change;
- protect forests from fire and insect outbreaks;
- protect forest soil organic matter by avoiding the use of broadcast slash burning and scarification treatments, preventing erosion, and minimizing soil disturbance (e.g., more logging on snow instead of in the summer);
- capture CO_2 emissions from pulp mills, kiln boilers, and other sources and channel them for beneficial uses that also remove CO_2, such as for growth enhancement in greenhouses;
- keep landfills anaerobic, and (or) tap their CH_4 emissions;
- promote longer rotations where tree species and sites are such that trees can be long-lived and large-statured (i.e., large C pools) with low risk of disturbance;
- use long-lived and potentially large-statured species to serve as C-reservoirs in long rotations, or very productive fast-growing species to quickly absorb C in short rotations;
- use fertilization where C-sequestration benefits exceed fertilization manufacture and application costs, especially if non-fossil fuel sources can be used (e.g., animal and human wastes);
- minimize regeneration delay and achieve high/full stocking as soon as possible;
- promote economic and policy instruments that encourage sustainable forest management practices, carbon storage/sequestration, and reduction in fossil fuel use.

Economic impacts of adaptation to climate change on forestry

Countries must be prepared to adapt to climate change because the Kyoto Protocol is but a small step forward in efforts to reduce atmospheric CO_2. Therefore, it is useful to examine briefly the possible impacts on the forestry sector and what adaptation will mean. Uncertainties in the underlying biological responses to the changing climate necessarily limit the robustness of the conclusions, and it is to hoped that in the coming years, there will be increasing effort to integrate the biogeochemical models with socio-economic ones. Some ecological projections of the effects of climate change on north-

Table 20.5. Rates of decay of forest ecosystem components (including wood products) upon harvest (data compiled from Hoen and Solberg 1994).

End-use category	Time (years) from felling until decay starts	Decay time (years) (from beginning of decay until all fibre has decomposed)
Bark in land fillings	0	8
Bark for burning	0	1
Needles	0	7–11
Branches, stumps, stems in forest	0	12
Root system after felling	0	100
Construction material	80	80
Furniture and interiors	20	50
Impregnated lumber	40	70
Pallets	2	23
Losses	0	1
Composites, plywood	17	33
Sawdust	1	2
Pulp/paper	1	2
Fuelwood	0	1

ern forestlands are reviewed earlier in this chapter, with more detailed analyses included in Apps et al. (1995), Apps and Price (1996), and in the series of publications associated with a conference on the role of boreal forests and forestry in the global carbon budget held May 2000, in Edmonton, Alberta (Apps et al. 2002; Karjalainen et al. 2002; Shaw and Apps 2002; Stocks et al. 2002).

Darwin et al. (1995) have provided a comprehensive model for analyzing the global response of primary sectors to climate change. The model gives some indication of the extent to which land use in Canada might be affected by climate change. The model consists of a geographic information system and a computable general equilibrium economic model of the global economy, with climate projections from four global circulation models (GCMs). Land-use potential in the model is based primarily on climatic factors (moisture and temperature) because these also affect soil formation. Of all the regions in the model, Canada stands to gain the most from climate change from an economic point of view. Canadian gross domestic product is projected to increase by an average of 2.2% across the four GCM scenarios, the largest increase for any region. Output of forest products will also increase (by 33%), although the forested land base will be reduced by an average of 7%. This is because in Darwin et al.'s (1995) model, cropland, pasture, and forestland are more productive under projected global warming, even in the absence of a CO_2-fertilization effect. Much of the increase in arable crop production in Canada comes from a northward shift in the western grain belt at the expense of boreal forest, and a shifting of the highly productive corn-belt in the United States into the eastern portion of the Canadian grain belt.

Adaptation of forest management to a changing climate will be required in order to ensure that forest sector returns are maximized. Examples of management adaptation include salvaging dying trees, vegetation control to help offset drought, replanting with more suitable species, and shifting processing capacity to areas where timber is relatively plentiful (Binkley and van Kooten 1994). In addition, as the frequency of pest outbreaks and forest fires increases, investment in the management of pests and fire is also expected to rise. As the northern fringe of the boreal forest shifts northward, tree planting can help to maintain forested area as land is lost to agriculture on the southern border.

The conclusion from economic research is that climate change is unlikely to bring about reductions in the global supply of primary commodities, including wood products (Schimmelpfenning et al. 1996; also IPCC 2001*b*, pp. 877–912). Rather, if economic markets and institutions are sufficiently flexible, landowners will make decisions to take advantage of changes, such as producing different crops or new crop varieties, adopting new management regimes (e.g., greater use of irrigation, or enhanced silviculture), or expanding activities to new areas. However, market failure, policy failure, and institutional barriers (see Chap. 7) to land-use change can be obstacles to enhancement of well-being in the face of climate change. Government regulation, for example, may be one impediment to adaptation. Policy failure would occur, for example, if the public authority required forest companies to reforest cutover lands on the boreal–grassland interface, thereby preventing their conversion to a more productive use such as pasture or cropland (van Kooten 1995; van Kooten et al. 2002). Market failure would occur if no account was taken of climate change impacts on recreation and other non-timber benefits of forests, particularly biodiversity. The high probability of climate change, coupled with this potential land-use inertia, calls into question the entire premise and feasibility of sustainable forest management as envisioned for a fixed land base (see Chap. 2).

Sohngen et al. (1999) addressed questions related to biodiversity and climate change. They linked ecosystem productivity and economic trade models to GCM output to demonstrate that climate change leads not only to greater wood fibre output (as predicted by Darwin et al. 1995), but also to greater protection of biodiversity. The reason for the enhanced status of biodiversity is that, although all forests become more productive as a result of CO_2 fertilization (according to this model, but see also the earlier section on CO_2 fertilization), there is also greater investment in plantation forests, also because of CO_2 fertilization. As plantation harvests increase, wood fibre prices fall and natural forests (those that harbour biodiversity) are less susceptible to logging. The area of natural forests that is not attractive for logging increases. On the other hand, Kirkup (2000) indicates that maintaining biodiversity in the face of climate change is more challenging. Solutions may require extensive biotic inventorying and monitoring, and the maintenance or restoration of landscape linkages (see Chap. 12) to facilitate the migration of countless unmanaged and unmonitored species.

Conclusions

The future carbon balance in the boreal forest will largely depend on the type and frequency of disturbances, changes in species composition, and alterations to the nutrient and moisture regimes under changing climate conditions. It will also depend on forest management practices that affect both the disturbance regime and nutrient status.

Projected climate change scenarios for the boreal forest generally predict warmer and somewhat drier conditions, and the disturbance patterns are also expected to change. Forest fire and insect outbreaks, for example, have historically been sensitive to climate and are expected to change considerably under global warming. The variations in climatic conditions in central Canada over the past 20 years appear to have resulted in higher rates of forest disturbance at a national scale. Regionally, the changes may be larger: the mean fire return interval for some sites in Alberta and Saskatchewan has been reported to be as low as 34 years (Larsen and MacDonald 1998). With more frequent disturbances, more of the stands are in younger age classes and initially release greater amounts of carbon to the atmosphere as their elevated detritus and soil pools decay. This situation is expected to worsen as climatic change proceeds. Altered boreal forest disturbance regimes — especially increases in frequency, size, and severity — may release carbon from vegetation, the forest floor, and soil at higher rates than the rate of carbon accumulation in regrowing vegetation.

The realization of the potential increase in plant productivity due to climate change will be determined by a variety of factors such as changes in species and competitive interactions, water availability, and the effect of temperature increase on photosynthesis and respiration. The precise balance of carbon uptake and release depends on the detailed processes, and especially interactions between climate, site variables, and vegetation over the changing life cycles of forest stands. Quantifying life-cycle dynamics at the stand level is essential for projecting future changes in forest-level carbon stocks. The net effect of such changes could result in positive feedback to climate change and thereby accelerate global warming (Kurz et al. 1995*b*).

Forest management options to enhance or protect carbon stocks include reducing the regeneration delay through seeding and planting, enhancing forest productivity, changing the harvest rotation length, the judicious use of forest products, and forest protection through control and suppression of disturbance by fire, pests, and disease. A protected forest acts as a sink only during the period of transition from a higher to a lower period of disturbance. Can such protection be maintained if the changing climate favours increased disturbances? Moreover, it must be stressed that if the protection is removed, the same areas will be a source as the disturbance regime relaxes back to a period of higher disturbance.

Policy options that nations can pursue to minimize their contributions to global climate change will include land management activities such as forest management and afforestation. Land management agencies and forest managers can choose from a large array of tools (see Box 20.4) in implementing carbon management strategies that will impact carbon stock changes and therefore net carbon removals and emissions.

Given the potential importance of terrestrial carbon sinks, carbon offsets can be an important element for encouraging economics investments in carbon sequestration projects, with verification by authorized certifiers. If CO_2-emission taxes are to be employed, then these need to be extended to cover carbon sinks (e.g., afforestation projects), although this will require that landowners be taxed for release of CO_2 to the atmosphere but subsidized for its removal (van Kooten et al. 1995). Tax revenues could be used to subsidize removals. The effectiveness of an efficient tax/subsidy system could be maximized by covering as broad a range of activities affecting carbon emissions as possible. This does not mean that carbon sink options will be undertaken on a large scale, however. Only if terrestrial carbon sinks are competitive with other means of reducing a nation's CO_2 emissions will private firms start paying for forest management and afforestation projects. To remain globally competitive, boreal jurisdictions must create as flexible an environment as possible, allowing private firms to seek out the most cost-effective options for reducing emissions.

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References

Adams, D.M., Alig, R.J., McCarl, B.A., Callaway, J.M., and Winnett, S.M. 1999. Minimum cost strategies for sequestering carbon in forests. Land Econ. **75**: 360–374.

Alm, J., Talanov, A., Saarnio, S., Silvola, J., Ikkonen, E., Aaltonen, H., Nykanen, H., and Martikainen, P.J. 1997. Reconstruction of the carbon balance for microsites in a boreal oligotrophic pine fen, Finland. Oecologia, **110**: 423–431.

Amiro, B.D., Stocks, B.J., Alexander, M.E., Flannigan, M.D., and Wotton, B.M. 2001*a*. Fire, climate change, carbon and fuel management in the Canadian boreal forest. Int. J. Wildland Fire, **10**: 405–413.

Amiro, B.D., Todd, J.B., Wotton, B.M., Logan, K.A., Flannigan, M.D., Stocks, B.J., Mason, J.A., Martell, D.L., and Hirsch, K.G. 2001*b*. Direct carbon emissions from Canadian forest fires, 1959 to 1999. Can. J. For. Res. **31**: 512–525.

Apps, M.J., and Price, D.T. (*Editors*). 1996. Forest ecosystems, forest management and the global carbon cycle. Springer-Verlag, Heidelberg, Germany. NATO ASI Ser. 1, Vol. 40. 452 p.

Apps, M.J., Kurz, W.A., Luxmoore, R.J., Nilsson, L.O., Sedjo, R.A., Schmidt, R., Simpson, L.G., and Vinson, T.S. 1993*a*. Boreal forests and tundra. Water Air Soil Pollut. **70**: 39–53.

Apps, M.J., Kurz, W.A., and Price, D.T. 1993*b*. Estimating carbon budgets of Canadian forest ecosystems using a national scale model. *In* Proceedings of the Workshop on Carbon Cycling in Boreal Forests and Subarctic Ecosystems. *Edited by* T. Vinson and T. Kolchugina. United States Environmental Protection Agency, Office of Research and Development, Washington, D.C. EPA/600R-93-084. pp. 241–250.

Apps, M.J., Price, D.T., and Wisniewski, J. (*Editors*). 1995. Boreal forests and global change. Kluwer Academic Publishers, Dordrecht, Netherlands. 564 p.

Apps, M.J., Kurz, W.A., Beukema, S.J., and Bhatti, J.S. 1999. Carbon budget of the Canadian forest product sector. Environ. Sci. Policy, **2**: 25–41.

Apps, M.J., Bhatti, J.S., Halliwell, D.H., Jiang, H., and Peng, C.H. 2000. Simulated carbon dynamics in the boreal forest of central Canada under uniform and random disturbance regimes. *In* Global climate change and cold region ecosystems. *Edited by* R. Lal, J.M. Kimble, and B.A. Stewart. Advances in Soil Science, Lewis Publishers, Boca Raton, Florida. pp. 107–122.

Apps, M.J., Karjalainen, T., Stocks, B.J., and Shaw, C.H. 2002. Foreword. The role of boreal forests and forestry in the global carbon budget. Can. J. For. Res. **32**: v–vi.

Armentano, T.V. 1980. Drainage of organic soils as a factor in the world carbon cycle. BioScience, **30**: 825–830.

Armentano, T.V., and Menges, E.S. 1986. Patterns of change in the carbon balance of organic soil – wetlands of the temperate zone. J. Ecol. **74**: 755–774.

Arseneault, D., and Payette, S. 1992. A postfire shift from lichen-spruce to lichen-tundra vegetation at treeline. Ecology, **73**: 1067–1081.

Auclair, A.N.D., and Thomas, B.C. 1993. Forest wildfires as a recent source of CO_2 at mid-high northern latitudes. Can. J. For. Res. **23**: 1528–1536.

Baldocchi, D., Kelliher, F.M., Black, T.A., and Jarvis, P.G. 2000. Climate and vegetation controls on boreal zone energy exchange. Global Change Biol. **6**(Suppl. 1): 69–83.

Bazzaz, F.A., Bassow, S.L., Berntson, G.M., and Thomas, S.C. 1996. Elevated CO_2 and terrestrial vegetation: implications for and beyond the global carbon budget. *In* Global change and terrestrial ecosystems. *Edited by* B. Walker and W. Steffen. Cambridge University Press, Cambridge, U.K. pp. 43–76.

Beilman, D.W., Vitt, D.H., and Halsey, L.A. 2001. Localized permafrost peatlands in western Canada: definitions, distributions, and degradation. Arct. Antarct. Alp. Res. **33**: 70–77.

Betts, R.A. 2000. Offset of the potential carbon sink from boreal forestation by decreases in surface albedo. Nature (London), **408**: 187–90.

Bhatti, J.S., and Apps, M.J. 2000. Carbon and nitrogen storage in upland boreal forests. *In* Global climate change and cold regions ecosystems. *Edited by* R. Lal, J.M. Kimble, and B.A. Stewart. Advances in Soil Science, Lewis Publishers, Boca Raton, Florida. pp 79–89.

Bhatti, J.S., Apps, M.J., and Jiang, H. 2001. Examining the carbon stocks of boreal forest ecosystems at stand and regional scales. *In* Assessment methods for soil carbon. *Edited by* R. Lal, J.M. Kimble, R.F. Follett, and B.A. Stewart. Lewis Publishers, Boca Raton, Florida. pp. 513–532.

Bhatti, J.S., Apps, M.J., and Jiang, H. 2002*a*. Influence of nutrients, disturbances and site conditions on carbon stocks along a boreal forest transect in central Canada. Plant Soil, **242**: 1–14.

Bhatti, J.S., Apps, M.J., and Tarnocai, C. 2002*b*. Estimates of soil organic carbon stocks in central Canada using three different approaches. Can. J. For. Res. **32**: 805–812.

Binkley, C.S., and van Kooten, G.C. 1994. Integrating climatic change and forests: economic and ecological assessments. Clim. Change, **28**: 91–110.

Binkley, C.S., Apps, M.J., Dixon, R.K., Kauppi, P.E., and Nilsson, L.O. 1997. Sequestering carbon in natural forests *In* Economics of carbon sequestration in forestry. *Edited by* R.A. Sedjo, R.N. Sampson, and J. Wisniewski. Crit. Rev. Environ. Sci. Technol. **27**: 23–45.

Black, T.A., den Hartog, G., Neumann, H.H., Blanken, P.D., Yang, P.C., Russell, C., Nesic, Z., Lee, X., Chen, S.G., Staebler, R., and Novak, M.D. 1996. Annual cycles of water vapour and carbon dioxide fluxes in and above a boreal aspen forest. Global Change Biol. **2**: 219–230.

Blanken, P.D., Black, T.A., Neumann, H.H., den Hartog, G., Yang, P.C., Nesic, Z., Staebler, R., Chen, W., and Novak, M.D. 1998. Turbulent flux measurements above and below the overstory of a boreal aspen forest. Boundary-Layer Meteorol. **89**: 109–140.

Bloomberg, W.J. 1962. Cytospora canker of poplars: factors influencing the development of the disease. Can. J. Bot. **40**: 1271–1280.

Bonan, G.B., Pollard, D., and Thompson, S.L. 1992. Effects of boreal forest vegetation on global climate. Nature (London), **359**: 716–718.

Botch, M.S., Kobak, K.I., Vinson, T.S., and Kolchugina, T.P. 1995. Carbon pools and accumulation in peatlands of the former Soviet Union. Global Biogeochem. Cycles, **9**: 37–46.

Bramryd, T. 1979. The conservation of peatlands as global carbon accumulators. *In* Proceedings of the International Symposium on Classification of Peat and Peatlands, 17–21 September 1979, Hyytiälä, Finland. International Peat Society, Helsinki, Finland. pp. 296–305.

Brandt, J.P., Knowles, K.R., Larson, R.M., Ono, H., and Walter, B.L. 1996. Forest insect and disease conditions in west-central Canada in 1995 and predictions for 1996. Can. For. Serv. Info. Rep. NOR-X-347. 53 p.

Callan, B.E. 1998. Diseases of *Populus* in British Columbia: a diagnostic manual. Pacific Forestry Centre, Canadian Forest Service, Natural Resources Canada, Victoria, British Columbia. 157 p.

Campbell, I.D., and McAndrews, J.H. 1993. Forest disequilibrium caused by rapid Little Ice Age cooling. Nature (London), **366**: 336–338.

Campbell, I.D., Campbell, C., Vitt, D.H., Kelker, D., Laird, L.D., Trew, D., Kotak, B., LeClair, D., and Bayley, S. 2000. A first estimate of organic carbon storage in Holocene lake sediments in Alberta, Canada. J. Paleolimnol. **24**: 395–400.

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

Carignan, R., and Steedman, R.J. 2000. Impacts of major watershed perturbations on aquatic ecosystems. Can. J. Fish. Aquat. Sci. **57**(Suppl. 2): 1–4.

Carignan, R., D'Arcy, P., and Lamontagne, S. 2000. Comparative impacts of fire and forest harvesting on water quality in Boreal Shield lakes. Can. J. Fish. Aquat. Sci. **57**(Suppl. 2): 105–117.

Carroll, A.L., Hudak, J., Meades, J.P., Power, J.M., Gillis, T., McNamee, P.J., Wedeles, C.H.R., and Sutherland, G.D. 1995. EHLDSS: a decision support system for management of the eastern hemlock looper. *In* Proceedings of Decision Support 2001, 13–15 Sepember 1994, Toronto, Ontario. American Society for Photogrammetry and Remote Sensing, Bethesda, Maryland. pp. 807–824.

Carter, M.R., Angers, D.A., Gregorich, E.G., Donald, R.G., Monreal, C.M., Voroney, R.P., Rounsevell, M.D.A., and Loveland, P.J. 1994. The agricultural management effects on carbon sequestration in Eastern Canada. *In* Soil responses to climate change. *Edited by* M.D.A. Rounsevell. Springer-Verlag, Berlin, Germany. pp. 193–196.

Caspersen, J.P., Pacala, S.W., Jenkins, J.C., Hurtt, G.C., Moorcroft, P.R., and Birdsey, R.A. 2000. Contributions of land-use history to carbon accumulation in U.S. forests. Science, **290**: 1148–1151.

Cerezke, H.F., and Volney. 1995. Forest insect pests in the north west region. *In* Forest insect pests in Canada. *Edited by* J.A. Armstrong and W.G.H. Ives. Canadian Forest Service, Ottawa, Ontario. pp. 59–72.

Chapin, F.S., III, McGuire, A.D., Randerson, J., Pielke, R., Sr., Baldocchi, D., Hobbie, S.E., Roulet, N., Eugster, W., Kasischke, E., Rastetter, E.B., Zimov, S.A., Oechel, W.A., and Running, S.W. 2000. Feedbacks from arctic and boreal ecosystems to climate. Global Change Biol. **6**(Suppl. 1): 211–223.

Chen, W.J., Black, T.A., Yang, P.C., Barr, A.G., Neumann, H.H., Nesic, Z., Novak, M.D., Eley, J., and Cuenca, R. 1999. Effects of climate variability on the annual carbon sequestration by a boreal aspen forest. Global Change Biol. **5**: 41–53.

Chen, W., Chen, J., and Cihlar, J. 2000. An integrated terrestrial ecosystem carbon-budget model based on changes in disturbance, climate, and atmospheric chemistry. Ecol. Model. **135**: 55–79.

Chomitz, K.M. 2000. Evaluating carbon offsets from forestry and energy projects: how do they compare? Working Paper No. 2357. Development Research Group, World Bank, Washington, D.C. Available at http://www.econ.worldbank.org/docs/1111.pdf [viewed 19 May 2003]. 25 p.

Clymo, R. 1984. The limits to peat bog growth. Philos. Trans. R. Soc. Lond. B Biol. Sci. **303**: 605–654.

Clymo, R.S., Turunen, J., and Tolonen, K. 1998. Carbon accumulation in peatland. Oikos, **81**: 368–388.

Cohen, S., and Miller, K. 2001. North America. *In* Climate change 2001: impacts, adaptation and vulnerability. Contribution of Working Group II to the Third Assessment Report of the Intergovernmental Panel on Climate Change. *Edited by* J.J. McCarthy, O.F. Canziani, N.A. Leary, D.J. Dokken, and K.S. White. Cambridge University Press, Cambridge, U.K. pp. 735–800.

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

(Curtis) Canadian Pulp and Paper Association. 2000. CPPA's discussion paper on climate change. Montreal, Quebec. Available at http://www.open.doors.cppa.ca. [viewed January 2000].

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

Curtis, P.S., Vogel, C.S., Wang, X., Pregitzer, K.S., Zak, D.R., Lussenhop, J., Kubiske, M., and Teeri, J.A. 2000. Gas exchange, leaf nitrogen, and growth efficiency of *Populus tremuloides* in a CO_2-enriched atmosphere. Ecol. Appl. **10**: 3–17.

Darwin, R., Tsigas, M., Lewandrowski, J., and Raneses, A. 1995. World agriculture and climate change: economic adaptations. USDA Economic Research Service, Washington, D.C. Agric. Econ. Rep. 703. 86 p.

Dean, W.E., and Gorham, E. 1998. Magnitude and significance of carbon burial in lakes, reservoirs, and peatlands. Geology, **26**: 535–538.

Dixon, H.R., and Dixon, P.A. 1991. Quaternary ecology: a paleoecological perspective. Chapman and Hall, New York. 242 p.

Dixon, R.K., and Krankina, O.N. 1993. Forest fires in Russia: carbon dioxide emissions to the atmosphere. Can. J. For. Res. **23**: 700–705.

Duchemin, E., Lucotte, M., Canuel, R., and Chamberland, A. 1995. Production of the greenhouse gases CH_4 and CO_2 by hydroelectric reservoirs in the boreal region. Global Biogeochem. Cycles, **9**: 529–540.

Eugster, W., Rouse, W.R., Pielke, R.A., Sr., McFadden, J.P, Baldocchi, D.D., Kittel, T.G.F., Chapin, F.S, III, Liston, G.E., Luigi Vidale, P., Vaganov, E., and Chambers, S. 2000. Land-atmosphere energy exchange in arctic tundra and boreal forest: available data and feedbacks to climate. Global Change Biol. **6**(Suppl 1): 84–115.

Farquhar, G.D. 1997. Carbon dioxide and vegetation. Science, **278**: 1411.

Farquhar, G.D., von Caemmerer, S., and Berry, J.A. 1980. A biochemical model of photosynthetic CO_2 fixation in leaves of C3 species. Planta, **149**: 78–90.

Field, C.B., Jackson, R.B., and Mooney, H.A. 1995. Stomatal responses to increased CO_2: implications from the plant to the global scale. Plant Cell Environ. **18**: 1214–1225.

Fitzsimmons, M. 2002. Effects of deforestation and reforestation on landscape spatial structure in boreal Saskatchewan, Canada. Can. J. For. Res. **32**: 843–851.

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

Flannigan, M.D., and Woodward, F.I. 1994. Red pine abundance: current climate control and responses to future warming. Can. J. For. Res. **24**: 1166–1175.

Flannigan, M., Wotton, M., Carcaillet, C., Richard, P., Campbell, I., and Bergeron, Y. 1998. Fire weather: past, present and future. *In* Proceedings of the III International Conference on Forest Fire Research/14th Conference on Fire and Forest Meteorology, 16–20 November 1998, Luso (Coimbra), Portugal. Vol. 1. *Edited by* D.X. Viegas. ADAI, University of Coimbra, Coimbra, Portugal. pp. 113–128.

Flannigan, M.D., Stocks, B.J., and Wotton, B.M. 2000. Climate change and forest fires. Sci. Total Environ. **262**: 221–229.

Flannigan, M.D., Campbell, I.D., Wotton, M., Carcaillet, C., Richard, P.J.H., and Bergeron, Y. 2001. Future fire in Canada's boreal forest: paleoecology, GCM and RCM results. Can. J. For. Res. **31**: 854–864.

Fleming, R.A. 2000. Climate change and insect disturbance regimes in Canada's boreal forests. World Resour. Rev. **12**: 520–555.

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

Fowler, D., Cape, J.N., Coyle, M., Flechard, C., Kuylenstierna, J., Hicks, K., Derwent, D., Johnson, C., and Stevenson, D. 1999. The global exposure of forests to air pollutants. Water Soil Air Pollut. **116**: 5–32.

France, R., Steedman, R., Lehmann, R., and Peters, R. 2000. Landscape modification of DOC concentration in boreal lakes: implications for UV-B sensitivity. Water Air Soil Pollut. **122**: 153–162.

(Gardner) Forest Sector Table. 1999. Options report: options for the forest sector to contribute to Canada's National Implementation Strategy for the Kyoto Protocol. National Climate Change Program, Canadian Forest Service, Ottawa, Ontario.

Gardner, R.H., Hargrove, W.W., Turner, M.G., and Romme, W.H. 1996. Climate change, disturbance and landscape dynamics. *In* Global change and terrestrial ecosystems. *Edited by* B. Walker and W. Steffen. Cambridge University Press, Cambridge, U.K. pp. 149–172.

Gennings, C., Molot, L.A., and Dillon, P.J. 2001. Enhanced photochemical loss of organic carbon in acidic waters. Biogeochemistry, **52**: 339–354.

Gitay, H., Brown, S., Easterling, W. E., Jallow, B., Antle, J., Apps, M.J., Beamish, R., Chapin, T., Cramer, W., Franji, J., Laine, J., Erda, L., Magnuson, J. J., Noble, I., Price, C., Prowse, T.D., Sirotenko, O., Root, T., Schulze, E.-D., Sohngen, B., and Soussana, J.-F. 2001. Chapter 5: Ecosystems and their services. *In* Climate change 2001: impacts, adaptation and vulnerability. Contribution of Working Group II to the Third Assessment Report of the Intergovernmental Panel on Climate Change. *Edited by* J.J. McCarthy, O.F. Canziani, N.A. Leary, D.J. Dokken, and K.S. White. Cambridge University Press, Cambridge, U.K. pp. 235–342.

(Goncalves) Government of Canada. 2001. Canada's third national report on climate change 2001: actions to meet commitments under the United Nations Framework Convention on Climate Change. Minister of Public Works and Government Services, Ottawa, Ontario. Available at http://www.climatechange.gc.ca/english/3nr/index.html [viewed 19 May 2003]. 256 p.

(Goncalves) Government of Canada. 2002. Climate change plan for Canada. Minister of Public Works and Government Services, Ottawa, Ontario. Available at http://www.climatechange.gc.ca/plan_for_canada/index.html [viewed 19 May 2003]. 67 p.

Goncalves, J.L.M., and Carlyle, J.C. 1994. Modelling the influence of moisture and temperature on net nitrogen mineralization in a forested sandy soil. Soil Biol. Biochem. **26**: 1557–1564.

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

Goulden, M.L., Wofsy, S.C., Harden, J.W., Trumbore, S.E., Crill, P.M., Gower, S.T., Fries, T., Daube, B.C., Fan, S.M., Sutton, D.J., Bazzaz, A., and Munger, J.W. 1998. Sensitivity of boreal forest carbon balance to soil thaw. Science, **279**: 214–217.

Gower, S.T., Vogel, J.G., Norman, J.M., Kucharik, C.J., Steele, S.J., and Stow, T.K. 1997. Carbon distribution and aboveground net primary production in aspen, jack pine, and black spruce stands in Saskachewan and Manitoba. Canada. J. Geophys. Res. **102**(D24): 29029–29041.

Halsey, L.A., Vitt, D.H., and Zoltai, S.C. 1995. Disequilibrium response of permafrost in boreal continental western Canada to climate change. Clim. Change, **30**: 57–73.

Harden, J.W., Trumbore, S.E., Stocks, B.J., Hirsch, A., Gower, S.T., O'Neill, K.P., and Kasischke E.S. 2000. The role of fire in the boreal carbon budget. Global Change Biol. **6**(Suppl. 1): 174–184.

Harris, L.D. 1984. The fragmented forest: island biogeography theory and the preservation of biotic diversity. University of Chicago Press, Chicago, Illinois. 211 p.

Hauer, G., Williamson, T., and Renner, M. 2001. Socioeconomic impacts and adaptive responses to climate change: a Canadian forest sector perspective. Can. For. Serv. Inf. Rep. NOR-X-373. 55 p.

Henri, C.J. 2001. Current status of carbon offset markets: Where do forests stand? Paper presented at the Western Forest Economists Annual Meeting, 8 May 2001, Welches, Oregon.

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

Hogg, E.H. 1997. Temporal scaling of moisture and the forest-grassland boundary in western Canada. Agric. For. Meteorol. **84**: 115–122.

Hogg, E.H. 1999. Simulation of interannual responses of trembling aspen stands to climatic variation and insect defoliation in western Canada. Ecol. Model. **114**: 175–193.

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

Houghton, R.A., Skole, D.L., Nobre, C.A., Hackler, J.L., Lawerence, K.T., and Chomentowski, W.H. 2000. Annual fluxes of carbon from deforestation and regrowth in the Brazilian Amazon. Nature (London), **403**: 301–304.

IGBP, Terrestrial Carbon Working Group, Steffan, W., Noble, I., Canadell, P., Apps, M.J., Schulze, E.-D., Jarvis, P.G., et al. 1998. The terrestrial carbon cycle: implications for the Kyoto Protocol. Science, **280**: 1393–1394.

(IPCC) Intergovernmental Panel on Climate Change. 1994. Radiative forcing of climate change and an evaluation of the IPCC IS92 emissions scenarios. *Edited by* J.T. Houghton, L.G. Meira Fhilho, J. Bruce, H. Lee, B.A. Callander, E. Haites, N. Harris, and K. Maskell. Cambridge University Press, Cambridge, U.K.

(IPCC) Intergovernmental Panel on Climate Change. 1996. Climate change 1995: Impacts, adaptations and mitigation of climate change: Scientific–technical analysis. *Edited by* R.T. Watson, M.C. Zinyowera, R.H. Moss, and D.J. Dokken. Cambridge University Press, Cambridge, U.K.

(IPCC) Intergovernmental Panel on Climate Change. 2000. Land use, land-use change, and forestry. *Edited by* R.T. Watson, I.R. Novel, N.H. Bolin, N.H. Ravindranath, D.J. Verardo, and D.J. Dokken. Cambridge University Press, Cambridge, U.K.

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

(IPCC) Intergovernmental Panel on Climate Change. 2001*b*. Climate change 2001: impacts, adaptation and vulnerability. Contribution of Working Group II to the Third Assessment Report of the Intergovernmental Panel on Climate Change. *Edited by* J.J. McCarthy, O.F. Canziani, N.A. Leary, D.J. Dokken, and K.S. White. Cambridge University Press, Cambridge, U.K.

(IPCC) Intergovernmental Panel on Climate Change. 2001*c*. Climate change 2001: Mitigation. Third assessment report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, U.K.

Isebrands, J.G., Dickson, R.E., Rebbeck, J., and Karnosky, D.F. 2000. Interacting effects of multiple stresses on growth and physiological processes in northern forests. *In* Responses of Northern U.S. forests to environmental change. *Edited by* R.E. Mickler, R.A. Birdsey, and J. Hom. Springer-Verlag, Berlin, Germany. pp. 149–180.

Jarvis, P.G., Massheder, J.M., Hale, S.E., Moncrieff, J.B., Rayment, M., and Scott, S.L. 1997. Seasonal variation of carbon dioxide, water vapour, and energy exchanges of a boreal black spruce forest. J. Geophys. Res. **102**(D24): 28953–28966.

Jenkinson, D.S., Adams, D.E., and Wild, A. 1991. Model estimate of CO_2 emissions from soil in response to global warming. Nature (London), **351**: 304–306.

Johnson, L.C., and Damman, A.W.H. 1991. Species-controlled Sphagnum decay on a south Swedish raised bog. Oikos, **61**: 234–242.

Karjalainen T., Apps, M.J., Stocks, B.J., and Shaw, C.H. 2002. Foreword: the role of boreal forests and forestry in the global carbon budget. For. Ecol. Manage. **169**: 1–2.

Kasischke, E.S. 2000. Effects of climate change and fire on carbon storage in North American boreal forests. *In* Fire, climate change, and carbon cycling in the North American boreal forest. *Edited by* E.S. Kasischke and B.J. Stocks. Springer-Verlag, New York, New York. pp. 440–452.

Kasischke, E.S., O'Neill, K.P., French, N.H.F., and Bourgeau-Chavez, L.L. 2000. Controls on patterns of biomass burning in Alaskan boreal forests. *In* Fire, climate change, and carbon cycling in the North American boreal forest. *Edited by* E.S. Kasischke and B.J. Stocks. Springer-Verlag, New York, New York. pp. 173–196.

Kauppi, P.E., Sedjo, R.A., Apps, M.J., Cerri, C.C., Fujimori, T., Janzen, H., Krankina, O.N., Makundi, W., Marland, G., Masera, O., Nabuurs, G.J., Razali, W., and Ravindranath, N.H. 2001. Chapter 4: Technological and economic potential of options to enhance, maintain, and manage biological carbon reservoirs and geo-engineering. *In* The Intergovernmental Panel on Climate Change (IPCC) Working Group

III Contribution to the Third Assessment Report on Mitigation of Climate Change. *Edited by* B. Metz, O. Davidson, R. Swart, and J. Pan. Cambridge University Press, Cambridge, U.K. pp. 301–323.

Kelly, C.A., Rudd, J.W.M., Bodaly, R.A., Roulet, N.P., St. Louis, V.L., Heyes, A., Moore, T.R., Schiff, S., Aravena, R., Scott, K.J., Dyck, B., Harris, R., Warner, B., and Edwards, G. 1997. Increases in fluxes of greenhouse gases and methyl mercury following flooding of an experimental reservoir. Environ. Sci. Technol. **31**: 1334–1344.

Kimmins, J.P. 1992. Balancing act: environmental issues in forestry. UBC Press, Vancouver, British Columbia. 244 p.

Kirkup, P. 2000. Impacts of global change on biodiversity. Available at http://www.glg.ed.ac.uk/home/Paul.Kirkup/specmod/atmosessay.html [updated 17 January 2001; viewed 1 June 2002].

Kirschbaum, M.U.F., and Fishlin, A. 1996. Climate change impacts on forests. *In* Climate change 1995: Contributions of Working Group II to the Second Assessment Report of the Intergovernmental Panel on Climate Change. *Edited by* R. Watson, M.C. Zinowera, and R.H. Moss. Cambridge University Press, Cambridge, U.K. pp. 93–129.

Kobak, K.I., Kondrasheva, N.Y., and Turchinovich, I.E. 1998. Changes in carbon pools of peatland and forests in northwestern Russia during the Holocene. Global Planet. Change, **16–17**: 75–84.

Körner, C. 2000. Biosphere responses to CO_2-enrichment. Ecol. Appl. **10**: 1590–1619.

Körner, C., and Bazzaz, F.A. (*Editors*). 1996. Carbon dioxide, populations and communities. Academic Press, San Diego, California. 465 p.

Krcmar, E., and van Kooten, G.C. 2001. Timber, carbon uptake and structural diversity tradeoffs in forest management. FEPA Working Paper. University of British Columbia, Vancouver, British Columbia.

Krcmar, E., Stennes, B., van Kooten, G.C., and Vertinsky, I. 2001. Carbon sequestration and land management under uncertainty. Eur. J. Oper. Res. **135**: 616–629.

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

Kurz, W.A., Apps, M.J., Webb, T.M., and McNamee, P.J. 1992. The carbon budget of the Canadian forest sector: phase 1. Can. For. Serv. Inf. Rep. NOR-X-326. 93 p.

Kurz, W.A., Apps, M.J., Beukema, S. J., and Lekstrum, T. 1995*a*. 20th century carbon budget of Canadian forests. Tellus, **47B**:170–177.

Kurz, W.A., Apps, M.J., Stocks, B.J., and Volney, W.J. 1995*b*. Global climate change: disturbance regimes and biospheric feedbacks of temperate and boreal forests. *In* Biotic feedbacks in the global climate system: will the warming speed the warming? *Edited by* G. Woodwell. Oxford University Press, New York, New York. pp. 119–133.

Kurz, W.A., Beukema, S.J., and Apps, M.J. 1998. Carbon budget implications of the transition from natural to managed disturbance regimes in forest landscapes. Mitigation Adapt. Strategies Global Change, **2**: 405–421.

Lafleur, P.M., Roulet, N.T., and Admiral, S.W. 2001. Annual cycle of CO_2 exchange at a bog peatland. J. Geophys. Res. **106**: 3071–3081.

Larocque, I., Bergeron, Y., Campbell, I.D., and Bradshaw, R.H.W. 2003. Distribution of Thuja occidentalis in the southern Canadian boreal forest: long-term effect of local fire disturbance. J. Veg. Sci. In press.

Larsen, C.P.S., and MacDonald, G.M. 1998. Fire and vegetation dynamics in a jack pine and black spruce forest reconstruction using fossil pollen and charcoal. J. Ecol. **86**: 815–828.

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

Li, Z., Apps, M.J., Banfield, E, and Kurz, W.A. 2002. Estimating net primary production of forests in the Canadian Prairie Province using an inventory-based carbon budget model. Can. J. For. Res. **32**: 161–169.

Li, Z., Apps, M.J., Kurz, W.A., and Beukema, S.J. 2003. Belowground biomass dynamics in the Carbon Budget Model of the Canadian Forest Sector: recent improvements and implications for the estimation of NPP and NEP. Can. J. For. Res. **33**: 126–136.

Likens, G.E., Driscoll, C.T., and Buso, D.C. 1996. Long-term effects of acid rain: response and recovery of a forest ecosystem. Science, **272**: 244–248.

Lindroth, A., Grelle, A., and Moren, A.S. 1998. Long-term measurements of boreal forest carbon balance reveal large temperature sensitivity. Global Change Biol. **4**: 443–450.

Liski, J., Pussinen, A., Pingoud, K., Mäkipää, R., and Karjalainen, T. 2001. Which rotation length is favourable for carbon sequestration? Can. J. For. Res. **31**: 2004–2013.

Loehle, C. 1998. Height growth rate tradeoffs determine northern and southern range limits for trees. J. Biogeogr. **25**: 735–742.

Lovett, G. M. 1994 Atmospheric deposition of nutrients and pollutants in North America: An ecological perspective. Ecol. Appl. **4**: 629–650.

Luo, Y., White, L.W., Canadell, J.G., DeLucia, E.H., Ellsworth, D.S., Finzi, A., Lichter, J., and Schlesinger, W.E. 2003. Sustainability of terrestrial carbon sequestration: a case study in Duke Forest with inversion approach. Global Biogeochem. Cycles. In press.

Mäkilä, M. 1997. Holocene lateral expansion, peat growth, and carbon accumulation on Haukkasuo, a raised bog in southeastern Finland. Boreas, **26**: 1–14.

Mäkipää, R., Karjalainen, T., Pussinen, A., and Kellomäki, S. 1999. Effects of climate change and nitrogen deposition on the carbon sequestration of a forest ecosystem in the boreal zone. Can. J. For. Res. **29**: 1490–1501.

Mallett, K.I., and Volney, W.J.A. 1999. The effect of Armillaria root disease on lodgepole pine tree growth. Can. J. For. Res. **29**: 252–259.

Malmer, N. 1986. Vegetational gradients in relation to environmental conditions in northwestern European mires. Can. J. Bot. **64**: 375–383.

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

McCaughey, J.H., Lafleur, P.M., Joiner, D.W., Bartlett, P.A., Costello, A.M., Jelinski, D.E., and Ryan, M.G. 1997. Magnitudes and seasonal patterns of energy, water, and carbon exchanges at a boreal young jack pine forest in the BOREAS northern study area. J. Geophys. Res. **102**(D24): 28997–29007.

McEachern, P., Prepas E.E., Gibson J.J., and Dinsmore W.P. 2000. Forest fire induced impacts on phosphorus, nitrogen, and chlorophyll a concentrations in boreal subarctic lakes of northern Alberta. Can. J. Fish. Aquat. Sci. **57**: 73–81.

McLaughlin, S., and Percy, K. 1999. Forest health in North America: some perspectives on actual and potential roles of climate and air pollution. Water Air Soil Pollut. **116**: 151–197.

Mercier, G. 1998. Canada country study: climate impacts and adaptations — energy sector. In The Canada country study: climate impacts and adaptations. Vol. 7. National Sectoral Volume. Edited by G. Koshida and W. Avis. Environmental Adaptation Research Group, Environment Canada, Toronto, Ontario. pp. 383–404.

Micales, J.A., and Skog, K.E. 1997. The decomposition of forest products in landfills. Int. Biodeterior. Biodegrad. **39**: 145–158.

Minkkinen, K., and Laine, J. 1998. Long-term effect of forest drainage on the peat carbon stores of pine mires in Finland. Can. J. For. Res. **28**: 1267–1275.

Molot, L.A., and Dillon, P.J. 1996. Storage of terrestrial carbon in boreal lake sediments and evasion to the atmosphere. Global Biogeochem. Cycles, **10**: 483–492.

Moore, T.R., Roulet, N.T., and Waddington, J.M. 1998. Uncertainty in predicting the effect of climatic change on the carbon cycling of Canadian peatlands. Clim. Change, **40**: 229–245.

Munn, R.E., and Maarouf, A.R. 1997. Atmospheric issues in Canada. Sci. Tot. Environ. **203**: 1–11.

Nadelhoffer, K.J., Emmett, B.A., Gundersen, P., Kjønaas, O.J., Koopmans, C.J., Schleppi, P., Tietema, A., and Wright, R.F. 1999. Nitrogen deposition makes a minor contribution to carbon sequestration in temperate forests. Nature (London), **398**: 145–148.

Neilson, R.P. 1993. Vegetation redistribution: a possible biosphere source of CO_2 during climatic change. Water Air Soil Pollut. **70**: 659–673.

Nohrstedt, H. 2001. Response of coniferous forest ecosystems on mineral soils to nutrient additions: a review of Swedish experiences. Scand. J. For. Res. **16**: 555–573.

Norby, R.J., Wullschlenger, S.D., Gunderson, C.A., Johnson, D.W., and Ceulemans, R. 1999. Tree responses to rising CO_2 in field experiments: implications for the future forest. Plant Cell Environ. **22**: 683–714.

(NRCan) Natural Resources Canada. 1998. Canada's energy outlook: 1996–2020. Natural Resources Canada, Ottawa, Ontario. Available at http://www.nrcan.gc.ca/es/ceo/toc-96E.html [viewed 18 April 2002].

(NRCan) Natural Resources Canada. 1999*a*. Forest health in Canada: an overview 1998. Forest Health Network, Canadian Forest Service, Natural Resources Canada, Ottawa, Ontario.

(NRCan) Natural Resources Canada. 1999*b*. Canada's emissions outlook: an update. Analysis and Modelling Group, National Climate Change Process, Natural Resources Canada, Ottawa, Ontario.

(NWWG) National Wetlands Working Group. 1988. Wetlands of Canada. Polyscience Publications, Inc., Montreal, Quebec. Ecol. Land Class. Ser. 24. 452 p.

Obersteiner, M., Rametsteiner, E., and Nilsson, S. 2001. Cap management for LULUCF options: an economic mechanism design to preserve the environmental and social integrity of forest related LULUCF activities under the Kyoto Protocol. International Institute for Applied Systems Analysis, Laxenburg, Austria. Interim Rep. IR-01-011. Available at http://www.iiasa. ac.at/Publications/Documents/IR-01-011.pdf [viewed 19 May 2003]. 17 p.

Olsen, K., Collas, P., Boileau, P., Blain, D., Ha, C., Henderson, L., Liang, C., McKibbon, S., and Moel-a-l'Huissier, L. 2002. Canada's greenhouse gas inventory 1990–2000. Greenhouse Gas Division, Environment Canada. Ottawa, Ontario.

Olsen, R., Ellsworth, D.S., Johnsen, K.H., Phillips, N., Ewers, B.E., Maier, C., Schafer, K.V.R., McCarthy, H., Hendrey, G., McNulty, S.G., and Katul, G.G. 2001. Soil fertility limits carbon sequestration by forest ecosystem in a CO_2-enriched atmosphere. Nature (London), **411**: 469–472.

Ouimet, R., and Camire, C. 1995. Foliar deficiencies of sugar maple stands associated with soil cation imbalances in the Quebec Appalachians. Can. J. Soil Sci. **75**: 169–175.

Ovenden, L. 1990. Peat accumulation in northern wetlands. Quat. Res. **33**: 377–386.

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

Percy, K., Bucher, J., Cape, J., Ferretti, M., Heath, R., Jones, H.E., Karnosky, D., Matyssek, R., Muller-Starck, G., Paoletti, E., Rosengren-Brinck, U., Sheppard, L., Skelly, J., and Weetman, G. 1999. State of science and knowledge gaps with respect to air pollution impacts on forests: reports from concurrent IUFRO 7.04.00 working party sessions. Water Air Soil Pollut. **116**: 443–448.

Peterjohn, W.T., Melillo, J.M., Steudler, P.A., Newkirk, K.M., Bowles, F.P., and Aber, J.D. 1995. Responses of trace gas fluxes and N availability to experimentally elevated soil temperatures. Ecol. Appl. **4**: 617–625.

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

Prepas, E.E, Pinel-Alloul, B., Planas, D., Methot, G, Paquet, S, and Reedyk, S. 2001. Forest harvest impacts on water quality and aquatic biota on the Boreal Plain: introduction to the TROLS Lake Program. Can. J. Fish. Aquat. Sci. **58**: 421–436.

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

Rapalee, G., Trumbore, S.E., Davidson, E.A., Harden, J.W., and Velduis, H. 1998. Soil carbon stocks and their rates of accumulation and loss in a boreal forest landscape. Global Biogeochem. Cycles, **12**: 687–701.

Reader, R.J., and Stewart, J.M. 1972. The relationship between net primary production and accumulation for a peatland in southeastern Manitoba. Ecology, **53**: 1024–1037.

Richter, D.D., O'Neill, P., and Kasischke, E.S. 2000. Post-fire stimulation of microbial decomposition in black spruce forest soils: a hypothesis. *In* Fire, climate change, and carbon cycling in the North American boreal forest. *Edited by* E.S. Kasischke and B.J. Stocks. Springer-Verlag, New York, New York. pp. 197–313.

Rivers, J.S., Siegel, D.I., Chasar, L.S., Chanton, J.P., Glaser, P.H., Roulet, N.T., and McKenzie, J.M. 1998. A stochastic appraisal of the annual carbon budget of a large circumboreal peatland, Rapid River Watershed, northern Minnesota. Global Biogeochem. Cycles, **12**: 715–727.

Rizzo, B., and Wiken, E. 1992. Assessing the sensitivity of Canada's ecosystems to climate change. Clim. Change, **21**: 37–55.

Robinson, D.C.E., Kurz, W.A., and Pinkham, C. 1999. Estimating the carbon losses from deforestation in Canada. Prepared by ESSA Technologies Ltd., Vancouver, British Columbia, for the National Climate Change Secretariat, Ottawa, Ontario. Available at http://www.nccp. ca/NCCP/pdf/Deforest_Canada.pdf [viewed 19 May 2003]. 81 p.

Robinson, S.D., Park, J., and Kettles, I.M. 2001. Northern peatlands to monitor recent and future warming. Project A313, final report to the Climate Change Action Fund. Natural Resources Canada, Ottawa, Ontario.

Roland, J., Mackey, B.G., and Cooke, B. 1998. Effects of climate and forest structure on duration of forest tent caterpiller outbreaks across central Ontario, Canada. Can. Entomol. **130**: 703–714.

Roulet, N.T. 2000. Peatlands, carbon storage, greenhouse gases, and the Kyoto Protocol: prospects and significance for Canada. Wetlands, **20**: 605–615.

Safranyik, L., and Linton, D.A. 2002. Line transect sampling to estimate the density of lodgepole pine currently attacked by mountain pine beetle. Can. For. Serv. Inf. Rep. BC-X-392. 11 p.

Sandor, R.L., and Skees, J.R. 1999. Creating a market for carbon emissions opportunities for U.S. farmers. Choices, The Magazine of Food, Farm & Resource Issues. First Quarter. The American Agricultural Economics Association, Ames, Iowa. pp. 13–18.

Schimel, D.S. 1995. Terrestrial ecosystems and the carbon cycle. Global Change Biol. **1**: 77–91.

Schimel, D.S., House, J.I., Hibbard, K.A., Bousquet, P., Ciais, P., Peylin, P., Braswell, B.H., Apps, M.J., Baker, D., Bondeau, A., Canadell, J., Churkina, G., Cramer, W., Denning, A.S., Field, C.B., Friedlingstein, P., Goodale, C., Heimann, M., Houghton, R.A., Melillo, J.M., Moore, B., Murdiyarso, D., Noble, I., Pacala, S.W., Prentice, I.C., Raupach, M.R., Rayner, P.J., Scholes, R.J., Steffen, W.L., and Wirth, C. 2001. Recent patterns and mechanisms of carbon exchange by terrestrial ecosystems. Nature (London), **414**: 169–172.

Schimmelpfenning, D., Lewandrowski, J., Reilly, J., Tsigas, M., and Parry, I. 1996. Agricultural adaptation to climate change: issues of long-run sustainability. USDA Economic Research Service, Washington, D.C. Agric. Econ. Rep. 740. 57 p.

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

Schlamadinger, B., Apps, M., Bohlin, F., Gustavsson, L., Jungmeier, G., Marland, G., Pingoud, K., and Savolainen, I. 1997. Towards a standard methodology for greenhouse gas balances of bioenergy systems in comparison with fossil energy systems. Biomass Bioenergy, **13**: 359–375.

Schlentner, R.E., and Van Cleve, K. 1985. Relationship between CO_2 evolution from soil substrate temperature and substrate moisture in four mature forest types in interior Alaska. Can. J. For Res. **15**: 97–106.

Schneider, S.H. 1995. Foreword. *In* Boreal forests and global change. *Edited by* M.J. Apps, D.T. Price, and J. Wisniewski. Kluwer Academic Publishers, Boston, Massachusetts. pp. ix–xi.

Schroeder, W., and Kort, J. (*Editors*). 2001. Temperate agroforestry: adaptive and mitigative roles in a changing physical and socio-economic climate. Proceedings of the Seventh Biennial Conference on Agroforestry in North America and the Sixth Annual Conference of the Plains and Prairie Forestry Association, 12–15 August 2001, Regina, Saskatchewan. Association for Temperate Agroforestry, University of Missouri, Columbia, Missouri. 350 p.

Schulze, E.-D., Lloyd, J., Kelliher, F.M., Wirth, C., Rebmann, C., Lühker, B., Mund, M., Knohl, A., Milyukova, I., Schulze, W., Ziegler, W., Varlagin, A., Sogachov, A., Valentini, R., Dore, S., Grigoriev, S., Kolle, O., Tchebakova, N., and Vygodskaya, N. 1999. Productivity of forests in the Eurosiberian boreal region and their potential to act as a carbon sink — a synthesis. Global Change Biol. **5**: 703–722.

Sedjo, R.A., Wisniewski, J., Sample, A.V., and Kinsman, J.D. 1995. The economics of managing carbon via forestry: assessment of existing studies. Environ. Resour. Econ. **6**: 139–165.

Sellers, P.J., Hall, F.G., Kelly, R.D., et al. 1997. BOREAS in 1997: experiment overview, scientific results, and future directions. J. Geophys. Res. **102**: 28731–28770.

Shaw, C.H., and Apps, M.J. (*Editors*). 2002. The role of boreal forests and forestry in the global carbon cycle. Proceedings of IBFRA 2000 Conference, 8–12 May 2000, Edmonton, Alberta. Northern Forestry Centre, Canadian Forest Service, Edmonton, Alberta. 332 p.

Shorger, A.W. 1972. The passenger pigeon: its natural history and extinction. University of Oklahoma Press, Tulsa, Oklahoma. 424 p.

Sieben, B.G., Spittlehouse, D.L., Benton, R.A., and McLean, J.A. 1994. A first approximation of the effect of climate warming on the white pine weevil hazard in the Mackenzie River drainage basin. *In* Mackenzie Basin Impact Study (MBIS) interim report #2 — Proceedings of the Sixth Biennial AES/DIAND Meeting on Northern Climate and Mid-Study Workshop of the Mackenzie Basin Impact Study, 10–14 April 1994, Yellowknife, Northwest Territories. *Edited by* S.J. Cohen. Atmospheric Environment Services, Environment Canada, Downsview, Ontario. pp. 316–328.

Simpson, L.G., Botkin, D.B., and Nisbet, R.A. 1993. The potential aboveground carbon storage of North American forests. Water Air Soil Pollut. **70**: 197–205.

Smith, T.M., Cramer, W., Dixon, R.K., Leemans, R., Neilson, R.P., and Solomon, A.M. 1993. The global terrestrial carbon cycle. Water Air Soil Pollut. **70**: 19–37.

Sohngen, B., and Sedjo, R. 1999. Estimating potential leakage from regional forest carbon sequestration programs. RFF Working Paper. Resources for the Future, Washington, D.C. 26 p.

Sohngen, B., Mendelsohn, R., and Sedjo, R. 1999. Forest management, conservation and global timber markets. Am. J. Agric. Econ. **81**: 1–13.

Statistics Canada. 1997. Nonmetal mines, SCI 062 Catalogue 26-224X1B. Statistics Canada, Ottawa, Ontario.

Steele, S.J., Gower, S.T., Vogel, J.G., and Norman, J.M. 1997. Root mass, net primary production and turnover in aspen, jack pine, and black spruce forests in Saskatchewan and Manitoba, Canada. Tree Physiol. **17**: 577–587.

Stevenson, D.S., Johnson, C.E., Collins, W.J., Derwent, R.G., Shine, K.P., and Edwards, J.M. 1998. Evolution of tropospheric ozone radiative forcing. Geophys. Res. Lett. **25**: 3819–3822.

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

Stocks, B.J., Fosberg, M.A., Wotton, B.M., Lynham, T.J., and Ryan, K.C. 2000. Climate change and forest fire activity in North American boreal forests. *In* Fire, climate change, and carbon cycling in the

North American boreal forest. *Edited by* E.S. Kasischke and B.J. Stocks. Springer-Verlag, New York, New York. pp. 368–376.

Stocks, B.J., Apps, M.J., Karjalainen, T., and Shaw, C.H. 2002. Foreword. The role of boreal forests and forestry in the global carbon budget. Clim. Change, **55**: 1–4.

Stocks, B.J., Mason, J.A., Todd, J.B., Bosch, E.M., Wotton, B.M., Amiro, B.D., Flannigan, M.D., Hirsch, K.G., Logan, K.A., Martell, D.L., and Skinner, W.R. 2003. Large forest fires in Canada, 1959–1997. J. Geophys. Res. In press.

Szumigalski, A.R., and Bayley, S.E. 1997. Net above-ground primary production along a peatland gradient in central Alberta, Canada, in relation to environmental factors. Ecoscience, **4**: 385–393.

Thie, J. 1974. Distribution and thawing of permafrost in the southern part of the discontinuous permafrost zone in Manitoba. Arctic, **27**: 189–200.

Thompson, A.J., and Shrimpton, D.M. 1984. Weather associated with the start of mountain pine beetle outbreaks. Can J. For. Res. **14**: 255–258.

Thormann, M.N., and Bayley, S.E. 1997. Decomposition along a moderate-rich fen-marsh peatland gradient in boreal Alberta, Canada. Wetlands, **17**: 123–137.

Thormann, M.N., Szumigalski, A.R., and Bayley, S.E. 1999. Aboveground peat and carbon acumulation potentials along a bog-fen-marsh gradient in southern boreal Alberta, Canada. Wetlands, **19**: 305–317.

Townsend, A.R., Braswell, B.H., Holland, E.A., and Penner, J.E. 1996. Spatial and temporal patterns in terrestrial carbon storage due to deposition of anthropogenic nitrogen. Ecol. Appl. **6**: 806–814.

Trumbore, S.E., and Harden, J. 1997. Accumulation and turnover of carbon in organic and mineral soil of BOREAS northern study area. J. Geophys Res. **102**: 28817–28830.

Turetsky, M.R., Wieder, R.K., Williams, C.J., and Vitt, D.H. 2000. Organic matter accumulation, peat chemistry, and permafrost melting in peatlands of boreal Alberta. Ecoscience, **7**: 379–392.

Turetsky, M.R., and Wieder, R.K. 2001. A direct approach to quantifying organic matter lost as a result of peatland wildfire. Can. J. For. Res. **31**: 363–366.

Turetsky, M.R., Weider, R.K., Halsey, L.A., and Vitt, D.H. 2002. Current disturbance and the diminishing peatland C sink. Geophys. Res. Let. **29**(11): 1526–1526.

(Updegraff-ECE/FAO). United Nations Economic Commission for Europe and Food and Agriculture Organization. 2000. Forest products annual market review 1999–2000. Timber Bull. **53**(3): 23–37.

Updegraff, K., Pastor, J., Bridgham, S.D., and Johnston, C.A. 1995. Environmental and substrate controls over carbon and nitrogen mineralization in northern wetlands. Ecol. Appl. **5**: 151–163.

Valentine, D.W., Holland, E.A., and Schimel, D.S. 1994. Ecosystem and physiological controls over methane production in northern wetlands. J. Geophys. Res. **99**: 1563–1571.

van Kooten, G.C. 1995. Climatic change and Canada's boreal forest: socio-economic issues and implications for land use. Can. J. Agric. Econ. **43**: 133–148.

van Kooten, G.C. 2000. Economic dynamics of tree planting for carbon uptake on marginal agricultural lands. Can. J. Agric. Econ. **48**: 51–65.

van Kooten, G.C. 2002. A primer on the ecomonics of climate change. Working Paper. Department of Economics, University of Victoria, Victoria, British Columbia. 35 p.

van Kooten, G.C. 2003. A primer on the economics of climate change. Working Paper. March. Department of Economics, University of Victoria, Victoria, British Columbia. 83 p.

van Kooten, G.C., and Bulte, E.H. 2000. The economics of nature: managing biological assets. Blackwell Publishers, Oxford, U.K.

van Kooten, G.C., Thompson, W.A., and Vertinsky, I. 1993. Economics of reforestation in British Columbia when benefits of CO_2 reduction are taken into account. *In* Forestry and the environment: economic perspectives. *Edited by* W.L. Adamowicz, W. White, and W.E. Phillips. CAB International, Wallingford, U.K. pp. 227–247.

van Kooten, G.C., Binkley, C.S., and Delcourt, G. 1995. Effect of carbon taxes and subsidies on optimal forest rotation age and supply of carbon services. Am. J. Agric. Econ. **77**: 365–774.

van Kooten, G.C., Krcmar-Nozic, E., Stennes, B., and van Gorkom, R. 1999. Economics of fossil fuel substitution and wood product sinks when trees are planted to sequester carbon on agricultural lands in western Canada. Can. J. For. Res. **29**: 1669–78.

van Kooten, G.C., Stennes, B., Krcmar–Nozic, E., and van Gorkom, R. 2000. Economics of afforestation for carbon sequestration in western Canada. For. Chron. **76**: 165–72.

van Kooten, G.C., Shaikh, S., and Suchanek, S. 2002. Mitigating climate change by planting trees: the transaction costs trap. Land Econ. **78**: 559–572.

Viereck, L., and Dyrness, C.T. 1979. Ecological effects of Wiockersham Dome fire near Fairbanks, Alaska. USDA Forest Service, Portland, Oregon. Gen. Tech. Rep. PNW-90. 71 p.

Vitousek, P.M., Aber, J.D., Howarth, R.W., Likens, G.F., Matson, P.A., Schindler, D.W., Shlesinger, W.H., and Tilman, D. 1997. Human alternation of the global nitrogen cycle: sources and consequences. Ecol. Appl. **7**: 737–750.

Vitt, D.H., Halsey, L.A., and Zoltai, S.C. 2000. The changing landscape of Canada's western boreal forest: the current dynamics of permafrost. Can. J. For. Res. **30**: 283–287.

Volney, W.J.A., and Fleming, R.A. 2000. Climate change and impacts of boreal forest insects. Agric. Ecosyst. Environ. **82**: 283–294.

Waddington, J.M., and Price, J.S. 2000. Effect of peatland drainage, harvesting, and restoration on atmospheric water and carbon exchange. Phys. Geogr. **21**: 433–451.

Waddington, J.M., and Roulet, N.T. 2000. Carbon balance of a boreal patterned peatland. Global Change Biol. **6**: 87–97.

Wardle, D.A., Zackrisson, O., and Nilsson, M.C. 1998. The charcoal effect in boreal forests: mechanisms and ecological consequences. Oecologia, **115**: 419–426.

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

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

Williams, D.W., and Liebhold, A.M. 1997. Latitudinal shifts in spruce budworm (Lepidoptera: Tortricidae) outbreaks and spruce-fir forest distributions with climate change. Acta Phytopathol. Entomol. Hung. **32**: 205–215.

Wilson, B., van Kooten, G.C., Vertinsky, I., and Arthur, L.M. (*Editors*). 1999. Forest policy: international case studies. CABI Publishing, Wallingford, U.K. 656 p.

Wullschlenger, S.D., Post, W.M., and King, A.W. 1995. On the potential for a CO_2 fertilization effect in forests: estimates of the biotic growth factor based on 58 controlled-exposure studies. *In* Biotic feedbacks in the global climatic cycle. *Edited by* G.M. Woodwell and F.Y. McKenzie. Oxford University Press, Oxford, U.K. pp. 85–107.

Yavitt, J., and Lang, G. 1990. Methane production in contrasting wetland sites: response to organic-chemical components of peat and to sulfate reduction. Geomicrobiol. J. **8**: 27–46.

Yavitt, J.B., Williams, C.J., and Wieder, R.K. 1997. Production of methane and carbon dioxide in peatland ecosystems across North America: effects of temperature, aeration, and organic chemistry of peat. Geomicrobiol. J. **14**: 299–316.

Zoltai, S.C. 1972. Palsas and peat plateaus in central Manitoba and Saskatchewan. Can. J. For. Res. **2**: 291–302.

Zoltai, S.C., and Martikainen, P.J. 1996. The role of forested peatlands in the global carbon cycle. *In* Forest ecosystems, forest management and the global carbon cycle. *Edited by* M.J. Apps and D.T. Price. Springer-Verlag, Heidelberg, Germany. NATO ASI Ser. 1, Vol. 40. pp. 47–58.

Zoltai, S.C., Morrissey, L.A., Livingston, G.P., and de Groot, W.J. 1998. Effects of fire on carbon cycling in North American boreal peatlands. Environ. Rev. **6**: 13–24.