

# Carbon in Canada's boreal forest — A synthesis<sup>1</sup>

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**Abstract:** Canada's managed boreal forest, 54% of the nation's total boreal forest area, stores 28 Pg carbon (C) in biomass, dead organic matter, and soil pools. The net C balance is dominated by the difference of two large continuous fluxes: C uptake (net primary production) and release during decomposition (heterotrophic respiration). Additional releases of C can be high in years, or in areas, that experience large anthropogenic or natural disturbances. From 1990 to 2008, Canada's managed boreal forest has acted as C sink of 28 Tg C year<sup>-1</sup>, removing CO<sub>2</sub> from the atmosphere to replace the 17 Tg of C annually harvested and store an additional 11 Tg of C year<sup>-1</sup> in ecosystem C pools. A large fraction (57%) of the C harvested since 1990 remains stored in wood products and solid waste disposal sites in Canada and abroad, replacing C emitted from the decay or burning of wood harvested prior to 1990 and contributing to net increases in product and landfill C pools. Wood product use has reduced emissions in other sectors by substituting for emission-intensive products (concrete, steel). The C balance of the unmanaged boreal forest is currently unknown. The future C balance of the Canadian boreal forest will affect the global atmospheric C budget and influence the mitigation efforts required to attain atmospheric CO<sub>2</sub> stabilization targets. The single biggest threat to C stocks is human-caused climate change. Large C stocks have accumulated in the boreal because decomposition is limited by cold temperatures and often anoxic environments. Increases in temperatures and disturbance rates could result in a large net C source during the remainder of this century and beyond. Uncertainties about the impacts of global change remain high, but we emphasize the asymmetry of risk: sustained large-scale increases in productivity are unlikely to be of sufficient magnitude to offset higher emissions from increased disturbances and heterotrophic respiration. Reducing the uncertainties of the current and future C balance of Canada's 270 Mha of boreal forest requires addressing gaps in monitoring, observation, and quantification of forest C dynamics, with particular attention to 125 Mha of unmanaged boreal forest with extensive areas of deep organic soils, peatlands, and permafrost containing large quantities of C that are vulnerable to global warming.

*Key words:* CBM-CFS3, climate change, deforestation, disturbance, greenhouse gas, peatland.

**Résumé :** La forêt boréale aménagée du Canada, soit 54 % de la surface totale de forêt boréale du pays, stocke 28 Pg de carbone (C) sous forme de biomasse, de nécromasse et de réserves dans les sols. Le bilan net du C est dominé par les différences entre deux grands flux continus : le captage du C (production primaire nette) et l'émission au cours de la décomposition (respiration hétérotrophe). Les émissions de carbone peuvent être élevées certaines années ou à certains endroits où se produisent de grandes perturbations naturelles ou anthropiques. De 1990 à 2008, la forêt canadienne aménagée a servi de puits de C à la hauteur de 28 Tg C an<sup>-1</sup>, captant le CO<sub>2</sub> de l'atmosphère pour remplacer les 17 Tg de C récoltées annuellement et mettre en réserve additionnelle 11 Tg de C an<sup>-1</sup> dans l'écosystème. Une grande partie (57 %) du C récolté depuis 1990 demeure accumulée dans les produits ligneux et les sites de déchets solides au Canada et à l'étranger, remplaçant le C émis par la décomposition ou la récolte de bois de chauffage avant 1990, contribuant ainsi à l'augmentation nette de l'accumulation dans les produits et les réserves de C enfouies. L'utilisation de produits ligneux a réduit les émissions dans d'autres secteurs en remplaçant des produits à fortes émissions (béton, acier). On ne connaît pas actuellement le bilan du C dans les forêts boréales non aménagées. Le bilan futur de C dans la forêt boréale canadienne affectera l'équilibre atmosphérique global et influencera les efforts d'atténuation requis pour atteindre les cibles de stabilisation du CO<sub>2</sub> atmosphérique. La plus grande menace pour le stockage du CO<sub>2</sub> est le changement climatique causé par l'homme. Le froid et des conditions anoxiques ont permis l'accumulation de grandes quantités de C dans la forêt boréale en limitant la décomposition. La hausse des températures et des taux de décomposition pourrait conduire à une importante génération de C net au cours de ce siècle et au-delà. L'incertitude quant à l'impact du réchauffement global demeure élevée, mais nous insistons sur l'asymétrie du risque : les augmentations soutenues à grande échelle de la productivité ne sont pas susceptibles d'atteindre un ordre de grandeur suffisant pour contre balancer les émissions provenant d'une augmentation des perturbations et de la respiration hétérotrophe. Pour réduire l'incertitudes quant au bilan actuel et futur des 270 Mha de la forêt boréale canadienne, il faudrait un examen des carences dans le suivi, l'observation et la quantification des dynamiques du C forestier, avec une attention particulière aux 125 Mha de forêt boréale non aménagées avec de grandes surfaces couvertes de sols organiques profonds, de tourbières et de pergélisol contenant de grandes quantités de C vulnérables au réchauffement global. [Traduit par la Rédaction]

*Mots-clés :* CBM-CFS3, changement climatique, déforestation, perturbations, gaz à effet serre, tourbière.

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## 1. Introduction

Globally, carbon (C) uptake by forests has removed about one third of the annual anthropogenic fossil fuel C emissions from the atmosphere since 1990 (Pan et al. 2011a). While the anthropogenic emissions rate continued to increase (Peters et al. 2012), the airborne fraction of carbon dioxide (CO<sub>2</sub>), i.e., the proportion of the anthropogenic emissions remaining in the atmosphere, has remained nearly constant (Le Quéré et al. 2009). This constancy is being attributed to an increase in the C uptake by land carbon sinks post-1988 and particularly to a major land sink in the Northern Hemisphere (Sarmiento et al. 2010). Forests are the dominant C-absorbing vegetation in the Northern Hemisphere (Pan et al. 2011a); and understanding the processes contributing to the C exchanges between the biosphere and the atmosphere, the processes that lead to the net removal of atmospheric C, and their future responses to direct and indirect human impacts, is a prerequisite for the design of global climate change adaptation and mitigation strategies. If forests continue to act as C sinks, removing CO<sub>2</sub> from the atmosphere, it will be easier to attain atmospheric CO<sub>2</sub> stabilization targets. However, if forests and other terrestrial ecosystems become weaker sinks or large-scale net emitters of CO<sub>2</sub> (e.g., Schuur et al. 2009; Boisvenue and Running 2010; Metsaranta et al. 2011; Schuur and Abbott 2011), the benefits of emissions reduction strategies in other sectors of the world will be reduced or completely overwhelmed (Smith et al. 2011; Warren 2011).

In 2010, forests covered 34% of North America's land area and accounted for 17% of the global forest area (FAO 2010). Canada's boreal forest land area (including other wooded land) covers 309 Mha (Brandt et al. 2013), 21%–27% of the global boreal forests (Brandt 2009) and 8% of the world's forests. Quantifying the contribution of Canada's boreal forest to the global C balance is a key objective of carbon science in Canada (CCP 2011; Canadian Forest Service, Natural Resources Canada 2012). This review synthesizes for the Canadian boreal forest the current understanding of C stocks, C exchanges between forests and the atmosphere, estimates of the net C budget, and the factors that will affect future C dynamics. It also outlines the gaps in our knowledge and understanding of C cycling in the Canadian boreal forest.

This paper is one of a series compiling the state of knowledge on and the status of Canada's boreal zone. Although it stands alone, the present review is complementary to related information on climate change impacts (Price et al. 2013) and climate change mitigation options in the forest sector (Lemprière et al. 2013). General background on Canada's boreal forest zone is presented by Brandt et al. (2013).

## 2. Characteristics of the boreal forest C cycle

As with all forest ecosystems, the C balance of Canada's boreal forest is strongly affected by the net balance of two large fluxes: C uptake from the atmosphere through net primary production (NPP) and C release through decomposition (heterotrophic respiration,  $R_h$ ). In addition to the two continuous processes of growth and decomposition that occur in all forests, some forests are in some years affected by anthropogenic or natural disturbance events. Disturbances such as harvesting, fire or insect outbreaks alter the magnitude and composition of C stocks through tree mortality, transfers of C to the atmosphere, and transfers of harvested material out of the forest. Quantifying forest C budgets involves estimation of C dynamics over a specified area (e.g., stand- or landscape-level) for a specified period of time, usually a growing season or year, as described in the next two sections.

### 2.1. Stand-level C dynamics

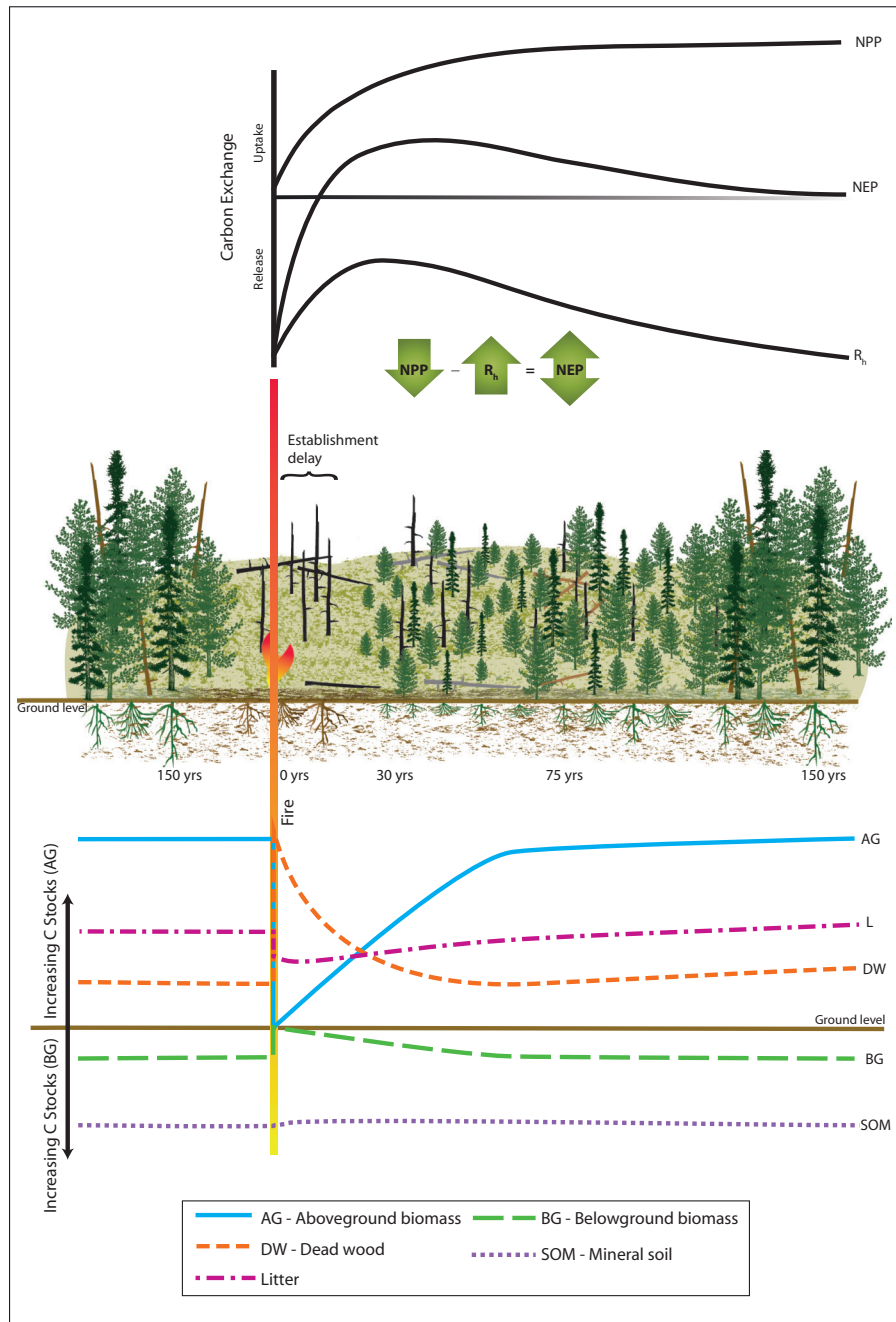
Carbon dioxide removed from the atmosphere through photosynthesis by trees and other plants is stored in forest ecosystems as C in above- and below-ground biomass, dead organic matter

(DOM; including litter, soil organic horizons, and standing or downed dead wood), and organic C in the mineral soil (IPCC 2003; Milakovskiy et al. 2012). Carbon is returned to the atmosphere through autotrophic respiration, decomposition, and oxidation in forest fires. The temporal dynamics of above- and below-ground biomass and hence C are linked because of the allometric relationships between tree size and root biomass (Kurz et al. 1996; Cairns et al. 1997; Li et al. 2003). The rate of photosynthesis (C uptake) is largely determined by site productivity (including climate), species composition, and vegetation age. Net primary production is a C flux defined as total photosynthesis (or gross primary production, GPP) minus the (autotrophic) respiration of primary producers (Chapin et al. 2006). Net ecosystem production (NEP) is estimated by subtracting  $R_h$  emissions (resulting from the decomposition of dead and soil organic matter) from NPP. It quantifies the accumulation (or loss) of C for a specified area (stand or a whole ecosystem) over a specified time (e.g., day, year, or decade). Net ecosystem exchange (NEE) refers to the C balance from the atmospheric perspective and thus has the reverse sign to NEP. NEE and NEP are related in that both include the difference between GPP and total ecosystem respiration, but NEE also includes direct emissions from fires and other nonrespiratory minor fluxes (Chapin et al. 2006, p. 1046). Over a set time period, if the ecosystem C balance results in a net uptake from the atmosphere (positive NEP), the ecosystem is said to be a C sink; in contrast, if ecosystem function results in net emissions to the atmosphere, the system is referred to as a source (negative NEP). The relative contribution of NPP and  $R_h$  to NEP changes over stand development (Fig. 1).

The net C balance with the atmosphere is periodically further affected by fluxes associated with disturbances. In boreal forests, stand-level C dynamics are characterized by long periods of slow C accumulation, punctuated by brief, often intense periods of disturbance (Fig. 1). Disturbance impacts range from reduction in growth rates for low severity disturbances to partial or complete stand replacement for more severe disturbances. Most disturbances result in a redistribution of C in the ecosystem including (1) transfer of live biomass to DOM from which C is slowly released through  $R_h$  over years to decades (e.g., Kurz et al. 2009), (2) oxidation and immediate release of some biomass and DOM C (e.g., Kasischke and Stocks 2000), and (3) transfer of C from the ecosystem to the forest product sector (Apps et al. 1999). Disturbance impacts can be described using disturbance matrices (Kurz et al. 1992) that quantify the amount of C in each predisturbance C pool that is redistributed to other pools, released to the atmosphere, or transferred out of the forest. Dead organic matter and soil C pools are typically larger after disturbance, decline for a period of years when decomposition losses are greater than input from turnover and mortality, and then increase again in older stands as this balance is reversed (Moroni 2006; Seedre et al. 2011; Deluca and Boisvenue 2012; Coursolle et al. 2012).

Following stand-replacing disturbances, boreal forests are C sources for a period of time until a C compensation point is reached where NEP becomes positive. A period of maximum C uptake (sink) then occurs at intermediate stand ages. The time required to reach the C compensation point and maximum C uptake rate depend on the type and intensity of the last disturbance, the amount of C transferred out of the ecosystem through oxidation or harvest, the amount and composition of DOM remaining after disturbance, the rate of tree regeneration, site productivity, ensuing stand density, and management effects on NPP and  $R_h$ . The amount of DOM remaining after disturbance (and hence subsequent  $R_h$  rates) and post-disturbance tree regeneration are key in determining when the C compensation point is reached (Fig. 1). This regeneration delay is influenced by disturbance intensity, site condition, the presence of a seed source, quality of seed bed, and competing vegetation. Species capable of vegetative reproduction can resume C uptake rapidly, whereas

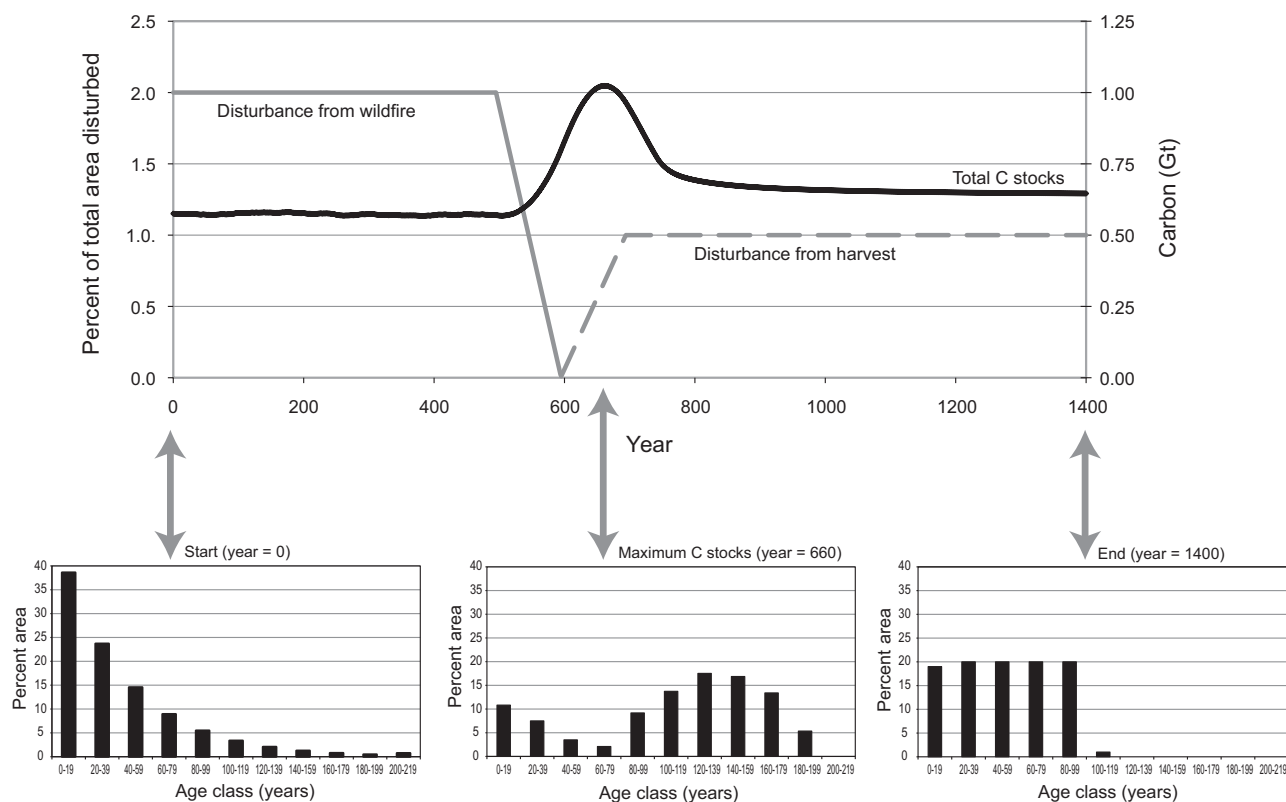
**Fig. 1.** Generalized schematic of the dynamics of stand-level net primary production (NPP), heterotrophic respiration ( $R_h$ , shown here as a negative flux and including in this graph large direct fire emissions in the year of the disturbance), and their net balance, net ecosystem production (NEP) (upper graph) following a fire disturbance. The generalized C stock dynamics following a fire disturbance are shown in the lower graph. For simplicity, dead belowground (coarse root) wood is included in the aboveground dead wood (DW) pool.



species regenerating from seed may have a longer C uptake delay. Site preparation, seeding, planting, and vegetation management can reduce regeneration delays and the time required to reach the C compensation point (Ryan et al. 2010). Boreal stands in Canada reach the C compensation point between 5 and 30 years after disturbance depending on disturbance type; 10–20 years post harvest, most boreal sites have reached the compensation point (Amiro et al. 2010; Grant et al. 2010; Coursolle et al. 2012). However,  $R_h$  fluxes from post-disturbance DOM can affect the net balance for much longer periods. For example, after a fire, dead standing trees initially decompose slowly until they fall over and through ground contact increase moisture levels and decomposi-

tion rates (Amiro et al. 2010; Seedre et al. 2011). In boreal forest stands, after a stand-replacing disturbance, NEP is lowest in the years immediately following the disturbance ( $-200 \text{ g C m}^{-2} \text{ year}^{-1}$ ) and reaches a maximum of  $210 \text{ g C m}^{-2} \text{ year}^{-1}$  for middle-aged stands (Amiro et al. 2010; CCP 2011). Net ecosystem production typically declines over time as NPP decreases with stand age due to reduced leaf area (Ryan et al. 1997; Smith and Long 2001) or changes in nutrient supply, respiration, C allocation or hydrological function (Battaglia 2001; Zaehle et al. 2006; Hember et al. 2012) (Fig. 1). Estimates for annual NEP averaged across all stand types and ages for the managed boreal forests of Canada between 1990 and 2008 ranged from  $-16.2$  to  $55.7 \text{ g C m}^{-2} \text{ year}^{-1}$ ,

**Fig. 2.** Simple demonstration of the impacts of changes in annual disturbance regime (frequency and type of disturbance) on landscape-level C storage. Changes in disturbance regimes (upper graph, left axis), resulting changes in landscape-level C stocks (upper graph, right axis) and forest age-class structure (lower graph), at the start point (left shifted), the point of maximum C density (transient right shifted), and the end point (normal age-class structure) of the simulations. Simulations conducted with the CBM-CFS3 using a hypothetical landscape with a single, typical boreal forest yield curve.



based on the data for the boreal zone extracted from Canada's National Forest Carbon Monitoring, Accounting, and Reporting System (NFCMARS; Stinson et al. 2011).

In boreal forests, a single stand can either be in a phase of high C uptake or high C storage but not both, as these phases occur at different times during stand development (Fig. 1). In very old stands, DOM pools such as coarse woody debris and litter are often the only pools that continue to increase in size, as biomass has reached or exceeds maximum C storage capacity. Ongoing transfers to DOM pools through biomass mortality and turnover allow these pools to accumulate C. While there has been debate about the role of old-growth forests as continuing C sinks (Luyssaert et al. 2008; Wirth et al. 2009), both forest inventory and flux tower measurements confirm that forest ecosystem C storage is finite (Gupta and Roa 1994; Zhang and Justice 2001; Laclau 2003; Keith et al. 2009, 2010). Depending on environmental conditions, old boreal forests can be C sources or small sinks (Dunn et al. 2007; Zha et al. 2009; Kurganova et al. 2010; Coursolle et al. 2012).

## 2.2. Landscape-level C dynamics

The Canadian boreal forest landscape consists of stands of different ages, disturbance histories, and species, growing on a range of site conditions. The landscape-level net C balance, or net biome production (NBP), can be calculated as the sum of the NEP of each component stand minus losses from disturbances to the atmosphere and transfers to the forest product sector (Kurz et al. 1992; IPCC 2003). At the stand scale, C losses due to disturbances are relatively infrequent and difficult to measure directly.

The landscape-level C balance is strongly affected by the proportion of area in different age classes (i.e., the age-class structure) due to the effect of age on stand-level NEP (Kurz et al. 1998;

Böttcher et al. 2008). The current age-class structure is a legacy of past disturbances and in turn affects the future C uptake rate (Kurz and Apps 1999; Rhemtulla et al. 2009a, 2009b; Pan et al. 2011b). Age-class structure effects can often explain observed differences in forest C stocks and fluxes in different regions.

When changes in disturbance regimes affect the age-class structure, they cause corresponding changes in landscape-level C dynamics (Fig. 2; Kurz and Apps 1999). An increasing annual disturbance rate lowers the average age of a landscape, resulting in reduced sinks or increased sources during the transition (Kurz et al. 1998; Kurz and Apps 1999; Boisvenue et al. 2012). A decreasing annual disturbance rate leads to an older landscape with increased sinks or reduced sources during the transition (Kurz and Apps 1999; Metsaranta et al. 2010, 2011). Recently disturbed landscapes with predominantly young forests are likely to be well below their C storage capacity, whereas very old landscapes with average ages higher than those expected from prevailing disturbance regimes may be above their sustainable C storage capacity. However, an old or right-shifted age-class structure, as currently observed in parts of the boreal zone of North America (Kurz and Apps 1999; Pan et al. 2011b), is a transient, nonsustainable phenomenon arising from a period with higher disturbance rates followed by a period with lower disturbance rates. Such age-class structures can result in temporary landscape-level C stocks in excess of a long-term sustainable maximum (Keith et al. 2010). As forested landscapes continue to age, disturbance susceptibility increases (Kurz et al. 2008b; Raffa et al. 2008), ultimately resulting in a period of increasing disturbance rates associated with increased C losses and a reduction in the average age of the landscape (e.g., Hély et al. 2010). Understanding and quantifying the relationships between current and future landscape-level C storage,

C dynamics, age-class structures, and disturbance regimes (Fig. 2) may provide opportunities for climate change mitigation activities in the forest sector (see Lemprière et al. 2013).

### 3. Current state of the boreal forest C cycle

This section reviews the methods used to determine C stocks and fluxes in boreal forest landscapes, provides estimates of the C budget for Canada's managed and unmanaged boreal forests, and discusses the potential contributions of processes that are currently not quantified in reported C estimates.

#### 3.1. Methodological considerations

Because of the complexity of processes involved in the C cycle and the scales at which these occur, stand and regional forest C balances cannot be measured directly. Methods have been developed that allow the estimation of forest C budgets using different data sources and principles. These include repeated measurements of forest characteristics, summations of C flux estimates, modelling of ecosystem processes, and atmospheric inversion models that allow inference of C fluxes from terrestrial and aquatic ecosystems over large regions. Carbon budget estimates derived from multiple lines of inquiry using independent data provide opportunities to assess uncertainties in these estimates.

##### 3.1.1. Field measurements

Site-level measurements of C balances provide the information required to evaluate or parameterize ecosystem models and for examining the magnitude and causes of interannual variability and trends in ecosystem production. Very few direct C flux measurements exist. Eddy covariance (EC) installations measure CO<sub>2</sub> (water and energy) exchange between the atmosphere and the forest and provide high temporal resolution estimates of net fluxes from all pools for relatively small areas (i.e., the tower footprint). However, even these measurements are incomplete and model-based gap-filling is used to produce complete time series (Barr et al. 2007; Moffat et al. 2007; Aubinet et al. 2012). Eddy covariance installations typically do not operate during periods of disturbance, and so do not capture short-term events that can cause very large C fluxes to the atmosphere (Körner 2003).

The stand-level C balance can also be estimated from biometric measurements of the size and turnover in ecosystem C pools made over annual or multiyear periods (Clark et al. 2001; Hoover 2008). The evolution of NEP over time closely follows the evolution of yields over time (Bernier et al. 2010; Kauppi et al. 2010) and can relatively easily be derived from yield curves and estimates of DOM and soil C dynamics (Kurz et al. 1992; IPCC 2003). The advantage of tree- or stand-level biometric measurements is that large numbers of these plots exist across Canada's boreal forest. They were established and are maintained by provincial and territorial resource management agencies, industry, and the National Forest Inventory (NFI; Gillis et al. 2005) to obtain data for developing and testing yield curves and growth models. Such data represent an important legacy of forest science (Zeide 2002) and are invaluable for developing models. They can be used to test the magnitude and evaluate the causes of hypothesized trends in productivity and mortality associated with global change (Hember et al. 2012). However, the temporal resolution of remeasurement (5–10 years) is too coarse to investigate interannual variability of growth in individual plots.

At annual timescales, biometric measurements do not systematically over- or under-estimate annual NEP derived from EC. Over multiyear time periods, biometric estimates of C stock changes and the sum of annual fluxes estimated by EC should converge, which is observed in some studies (Vedrova et al. 2006; Gough et al. 2008), but not in others (Curtis et al. 2002; Ohtsuka et al. 2009). Resources available for forest inventories and ecosystem-C-monitoring activities have been declining in Canada at a time

when global change impacts are affecting forest C dynamics, and yield curves will become increasingly unreliable (Hember et al. 2012; Huang et al. 2013). Ongoing monitoring of global change impacts on stand dynamics can contribute to reduced uncertainties and improve the quality of information used in forest planning by providing insights into forest regions with positive (Hember et al. 2012) or negative (Michaelian et al. 2011) responses to global changes.

Dendrochronological stand reconstruction methods that estimate both growth increment (e.g., Osawa et al. 2005) and mortality (Metsaranta et al. 2008) represent a third approach for obtaining spatially extensive annual resolution data on ecosystem productivity for several decades into the past (Graumlich et al. 1989; Metsaranta and Kurz 2012). These have the potential to expand the spatial and temporal coverage of annual forest productivity estimates, providing additional data for developing and testing ecosystem models (Metsaranta and Kurz 2012).

##### 3.1.2. Approaches to C balance estimations

Landscape-scale forest C balances can be estimated using a combination of measurements and modelling approaches ranging from allometric equations used to estimate tree-level biomass (Lambert et al. 2005; Ung et al. 2008) to forest ecosystem models used to estimate C budgets at biome, national, or global scales (Chen et al. 2000; Kurz et al. 2009; Boisvenue and Running 2010).

The Intergovernmental Panel on Climate Change (IPCC) describes two fundamentally different approaches for estimating forest C budgets (IPCC 2003). Method 1 (the default method) calculates the difference between annual gains (growth) and losses (mortality, disturbances, and harvesting); and method 2 (the stock change method) calculates the difference in total C stocks at two times (IPCC 2003). The first method is implemented in Canada and other countries using a single initial forest inventory and information on annual gains and losses (Stinson et al. 2011); whereas the second method is implemented in the US, Sweden, and other countries on repeated forest inventories (Heath et al. 2011; Swedish Environmental Protection Agency 2011). Reporting based on a single inventory and change information incorporates annual data on forest growth, decay, and disturbance to estimate annual C stock changes. This allows for separation of CO<sub>2</sub> and non-CO<sub>2</sub> greenhouse gas (GHG) emissions attributed to combustion, from CO<sub>2</sub> emissions attributed to decay. Carbon stock changes determined by the difference from successive inventories are potentially more accurate but cannot provide annual estimates for the period between inventories and require additional information to estimate emissions from non-CO<sub>2</sub> GHGs. Thus, the magnitude and cause of interannual variability cannot be determined from successive inventories alone. Moreover, where inventory estimates are based on annual partial remeasurements of only a subset of plots, the impacts of changing rates of natural disturbances, land-use change, and forest management are only observed with considerable delay (Bechtold and Patterson 2005). Over multiyear time periods, stock changes integrate the sum of annual fluxes over time and represent a standard against which estimates derived from other approaches can be assessed. Testing for convergence of the two methods has to date been limited to regional studies (Trofymow et al. 2008; Bernier et al. 2010), and even these were constrained because DOM and soil C stocks were not inventoried. A sample-based inventory of C stocks in Canada's entire boreal forest will be available from Canada's NFI (Gillis 2001; Gillis et al. 2005), but estimates of C stock changes will not be available until the ongoing first remeasurement is completed and analyzed.

Simulation models used in estimating C balances are usually composed of a series of empirical models, process representation, and steps to integrate knowledge and data from multiple sources. Forest growth in these models can be driven by yield information

(Kurz et al. 1992, 2009; Karjalainen et al. 2002; Schelhaas et al. 2004) or process representation (Chen et al. 2000; White et al. 2000; Zhou et al. 2008). Most models use process representation to account for DOM dynamics (White et al. 2000; Komarov et al. 2003; Liski et al. 2005; Kurz et al. 2009), but some landscape-level estimates of DOM stocks rely on statistical relationships between biomass and DOM and soil C (Smith and Heath 2002; Chen et al. 2010). Only some of the models account for the impacts of forest management, natural disturbances, or land-use change (McGuire et al. 2001; Kurz et al. 2009). The main advantage of using empirical field-based data originates from the large information content of inventories, which generally have good geographical coverage, acceptable temporal coverage, and are relatively easily available. This approach, however, carries the same limitations as empirically derived statistical estimation of growth (Monserud 2003) that include insensitivity to changing environmental conditions. In contrast, simulation models that rely on process representation respond to changing environmental inputs, but these models (e.g., Grant et al. 2010; Seidl et al. 2011) usually demand complex data for parameterization that are often difficult and costly to estimate. Further, many of the processes involved in C cycling are not yet well understood; for example, quantification of environmental impacts on growth and decomposition processes is still very uncertain as expressed in the disagreements among process models about the magnitude and sometimes even direction of environmental forcing of fluxes (Wang et al. 2011; Huntzinger et al. 2012). Most representations of soil processes in simulation models are over simplified because of the limited understanding of the processes (Conant et al. 2011; Schmidt et al. 2011) and the scarcity of data. Belowground processes account for a large portion of the uncertainty associated with current C budget estimates (Deluca and Boisvenue 2012).

Increasingly, simulation models incorporate widely available remote-sensing data. Worldwide estimates of productivity based on satellite information now exist (Zhao and Running 2010), as well as regional estimates using satellite data (Beck et al. 2011; Beck and Goetz 2011). Remote sensing estimates biomass C using methods that rely on reflectance-based variables such as vegetation indices (Hashimoto et al. 2012) or structural characteristics inferred from LiDAR or RADAR methods (Goetz et al. 2009; Wulder et al. 2012). To date, detection of forest disturbances from fires, insects, land use, and land-use change are the primary contributions of remotely sensed data to C estimation (Fraser et al. 2004; Leckie et al. 2006; de Groot et al. 2007; Wulder et al. 2008). However, recent advances in the understanding of the relationship between reflectance and forest physiology (Hilker et al. 2010), the detection of forest structure and forest biomass via remote sensing (Boudreau et al. 2008; Lefsky 2010; Wulder et al. 2012), and in satellite technology aimed at measuring C (NASA 2012b) may eventually yield improved estimates of C balances for areas where few field data are available. This will further increase the importance of remotely sensed data for the estimation of forest C balances.

In addition to inventory-based approaches and simulation models, atmospheric inversion models that allow inference of C fluxes from terrestrial and aquatic ecosystems over large regions can also be used to estimate regional-scale forest C fluxes (Michalak et al. 2004; Jacobson et al. 2007). The advantage of this approach is that it is built on a different foundation of knowledge and it employs nearly independent measurements from the inventory- and model-based approaches previously described. Because inversion models estimate C fluxes over an entire region, their estimates are not expected to be in close agreement with estimates obtained from forest carbon budget models in regions containing a matrix of forest, nonforest, and aquatic ecosystems (IPCC 2003). For example, wetlands, grasslands, croplands, other nonforest land categories, and aquatic surfaces contribute to the fluxes estimated by inversion models. In addition, there are still

substantial uncertainties among inversion estimates for the North America boreal region (McGuire et al. 2009; Hayes et al. 2012).

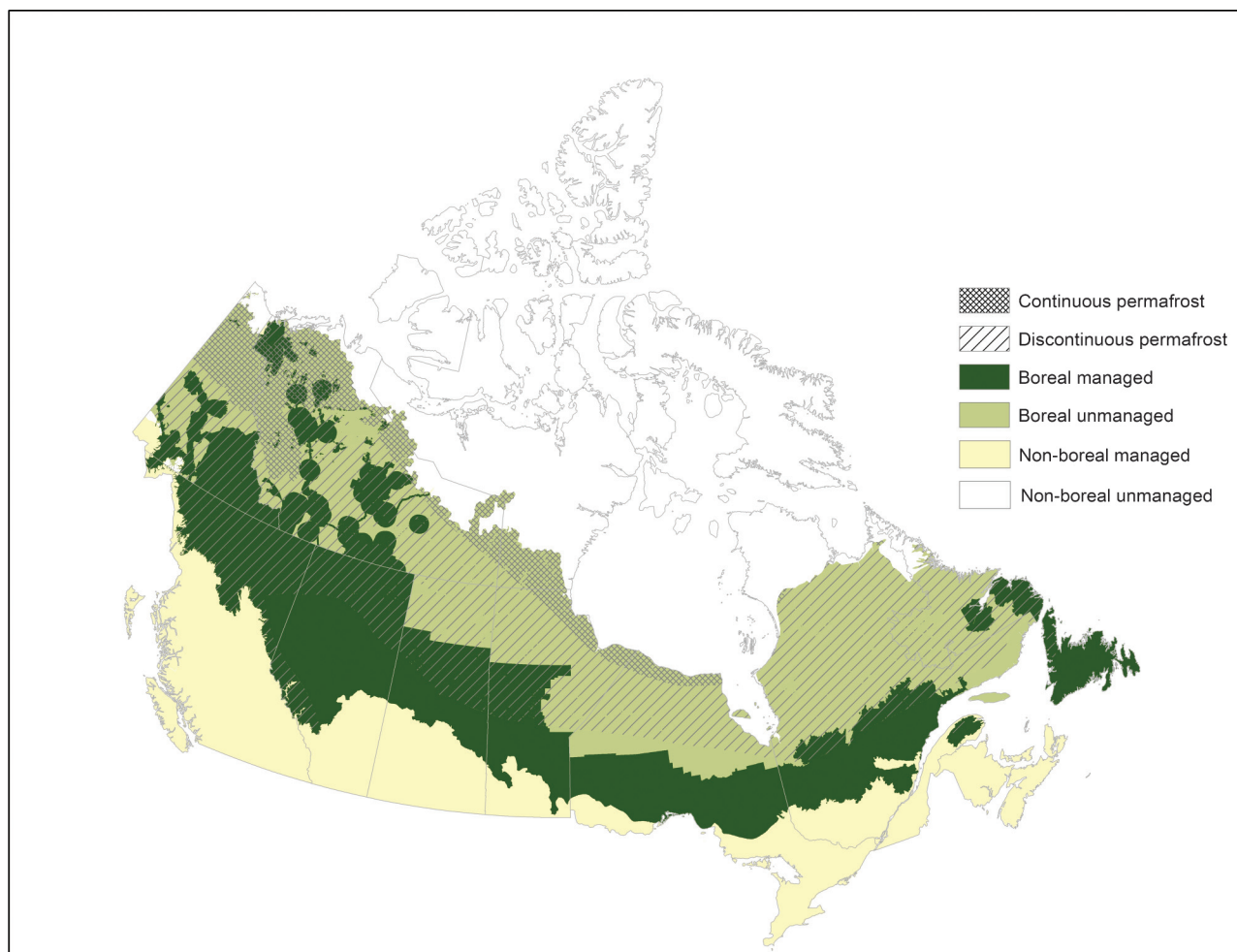
### 3.1.3. Canada's National Forest Carbon Monitoring, Accounting, and Reporting System

Estimates of the contemporary C budget of Canada's managed boreal forest that will be presented in section 3.2 are derived from Canada's NFCMARS. This system combines forest inventories, empirical yield tables developed from sample plot data, process-based modelling of DOM, and soil C dynamics, statistics on forest management activities, and remote sensing to estimate area, type, and location of natural disturbances and land-use change (Kurz and Apps 2006; Kurz et al. 2009; Stinson et al. 2011). It is used for monitoring (Canada's annual national GHG inventory report, Environment Canada 2009) and projection (e.g., forward-looking baselines in climate negotiations, Government of Canada 2011). It was also used in support of analyses of the election of forest management under Article 3.4 of the Kyoto Protocol (Kurz et al. 2008b) and is currently used for the assessment of climate change mitigation options in Canada's forest sector. The system and its core ecosystem model, the CBM-CFS3 (Kurz et al. 2009), were used in a number of scientific analyses at the regional (Kurz et al. 2008a, 2008c; Trofymow et al. 2008; Bernier et al. 2010; Metsaranta et al. 2011) and national scale (Kurz et al. 2008b; Metsaranta et al. 2010; Stinson et al. 2011). Canada's NFCMARS does not represent all potentially important processes, and the omission of some processes contributes to uncertainties in C budget estimates (see section 3.3).

### 3.2. Estimates of the contemporary C balance

Canada's boreal zone is generally understood to be acting as a weak net sink for atmospheric CO<sub>2</sub> though published studies differ with regard to the land surface fluxes included (Kurz and Apps 1999; Chen et al. 2000; Jacobson et al. 2007; Peters et al. 2007; McGuire et al. 2009; Ciais et al. 2010; Stinson et al. 2011; Hayes et al. 2012). The application of multiple lines of inquiry to estimate and attempt to understand the present contributions of Canada's boreal zone to the global C cycle provides important insights, both about the C balance itself and the limitations of current scientific knowledge. Atmospheric inversion modelling (often referred to as coming from a "top-down" perspective) and terrestrial ecosystem modelling (or "bottom-up" perspective) studies typically consider the North American boreal zone by accounting for the contributions of both forest and nonforest terrestrial ecosystems. However, it is difficult to derive estimates of the forest C balance specific to the Canadian boreal zone (Brandt 2009) from these studies. Here, we are able to provide estimates for the managed forest area within the boreal zone (Brandt 2009) from the inventory-based modelling estimates developed from Canada's NFCMARS (Stinson et al. 2011). Approximately half of the area of the Canadian boreal zone is forested, with the remaining area occupied by wetlands, primarily peatlands (49.0 Mha or 9% of the zone; Lempriere et al. 2013), and water (70 Mha or 13% of the zone; Brandt et al. 2013). A little over half of the forested area is managed (Fig. 3). For the purpose of estimating and reporting GHG emissions and removals, the managed forest includes (i) lands managed for the sustainable harvest of wood fibre (e.g., saw logs, pulp logs) or wood-based bioenergy; (ii) lands under intensive protection from natural disturbances (e.g., fire and insect suppression to protect forest resources); and (iii) protected areas, such as national and provincial parks that are managed to conserve forest ecological values (Stinson et al. 2011). The C balance of the unmanaged forest area is discussed separately and interpreted from relevant studies with a large scope that may include unmanaged forest (section 3.2.2) and nonforested areas such as peatlands (including bogs and fens).

**Fig. 3.** Geographic relationship between the boreal zone as defined by Brandt (2009), the managed forest as defined by Canada for the purposes of reporting to the UNFCCC (Stinson et al. 2011), and permafrost zones (Hegginbottom et al. 1995).



Natural disturbances and land-use change affect the C balance of both managed and unmanaged forest areas, but the C balance of the managed forest is also affected by forest management practices including harvesting and protection against natural disturbances. Natural disturbances are a main driver of C dynamics in the boreal zone; they cause immediate emissions, transfer biomass to DOM that will decompose over future decades, reset stand age, and potentially initiate succession to a new trajectory (section 2.1). Interannual and multidecadal changes in disturbance regimes are a key driver of forest C sources and sinks (Kurz and Apps 1999; section 2.2). Fire is the primary driver of forest dynamics in western boreal forest (Fig. 4; Stocks et al. 2003; Bond-Lamberty et al. 2007b). The C balance is further affected by periodic regional outbreaks of insects, including mountain pine beetle (*Dendroctonus ponderosae* Hopkins) (Kurz et al. 2008c), spruce budworm (*Choristoneura fumiferana* Clem.) (Kurz et al. 2008b; Dymond et al. 2010), and other insects. Other disturbances, such as windthrow (Lindroth et al. 2009) and ice storms (Olthof et al. 2004; Pisaric et al. 2008), are of limited regional significance in Canada's boreal forest. Land-use change (deforestation and conversion to nonforest land use) affects less than 0.02% of Canada's boreal forest annually (section 3.2.3).

Forest management affects the C budget of the boreal zone through harvesting using various techniques, site preparation (slash and soil treatments), planting, and suppression of fires and insects. Management is predominantly extensive, with limited silvicultural activity beyond timber harvesting, site preparation

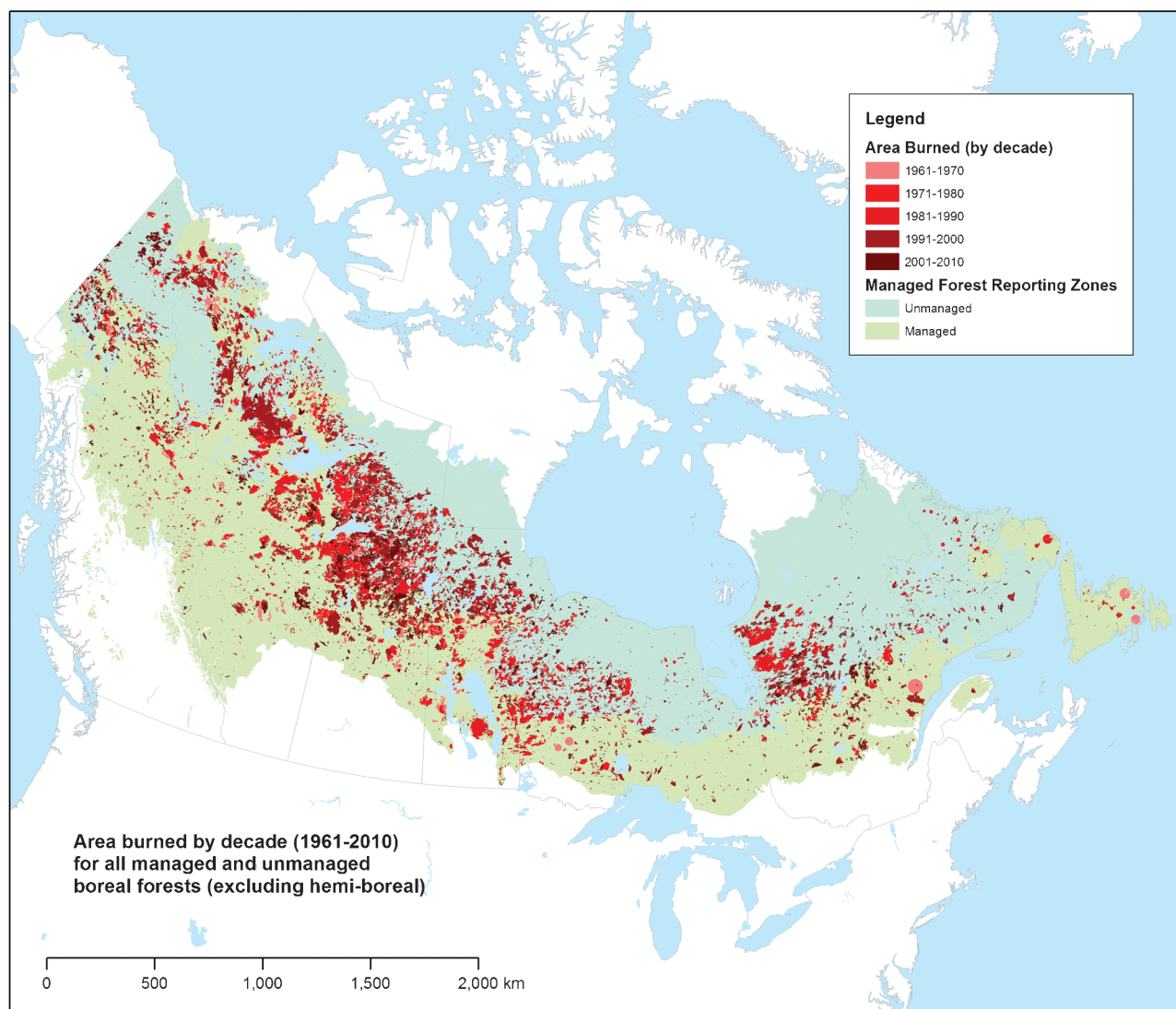
for artificial or natural regeneration, pre-commercial thinning to reduce competition, and some active stand re-establishment (e.g., tree planting, seeding) following harvesting (National Forestry Database 2012). Fire and insect disturbance suppression is practiced with the aim of balancing the need to control fires against the desire to allow these disturbances to play their ecologically beneficial role where possible (Duchesne and Hawkes 2000). Harvesting transfers C from the forest to society, providing timber, fibre, and biomass for energy use; and the contribution of harvested wood products to the C balance is discussed in section 3.2.4. Harvesting, like natural disturbances, also contributes to forest renewal and the maintenance of the forest sink strength.

### 3.2.1. Managed forest area in the boreal zone

Estimates of the C balance for managed forests within the boreal zone were derived from Canada's NFCMARS using the 1990–2008 time series described in Stinson et al. (2011). Of Canada's 270 Mha boreal forest area (does not include other wooded land; Brandt et al. 2013), 145 Mha are considered managed for the purpose of reporting GHG emissions and removals and 125 Mha are unmanaged (Fig. 3).

The managed forest area in the boreal zone (Fig. 5a) stores approximately 28 Pg C, mostly in the four boreal ecozones with minor contributions from other ecozones included in the definition of the boreal zone (Brandt 2009). The C density in the managed forest area is approximately 193 Mg·ha<sup>-1</sup>, and densities are

**Fig. 4.** Area burned by decade (1961–2010) for all managed and unmanaged boreal forests of Canada, derived from the Canadian forest service large-fires database and the National burn area composite.



slightly lower for boreal ecozones on the shield than on the plains or cordillera (Fig. 5a). The proportional distributions of C between the five pools defined by the IPCC (IPCC 2003) do not vary much between ecozones (Fig. 5b). Approximately 25% of total C stocks are found in the biomass (above- plus below-ground), with the remainder in the deadwood, litter, and soil organic matter pools (Fig. 5b). The largest proportion (40%) is found in the soil organic matter.

Our estimate for the aboveground biomass C density ( $\sim 40 \text{ Mg}\cdot\text{ha}^{-1}$ ) is somewhat higher than a previous sample-based estimate for the North American boreal zone (strata IV and XI,  $31 \text{ Mg}\cdot\text{ha}^{-1}$ ; Botkin and Simpson 1990). Regional estimates in Quebec for commercial forest of  $27.7\text{--}36.6 \text{ Mg}\cdot\text{ha}^{-1}$  (Boudreau et al. 2008),  $42 \text{ Mg}\cdot\text{ha}^{-1}$  (ground-plot based), and  $33 \text{ Mg}\cdot\text{ha}^{-1}$  (remote-sensing based) (Nelson et al. 2009) are similar to our estimate of  $35.5 \text{ Mg}\cdot\text{ha}^{-1}$  for the boreal shield east. No other measurement-based regional- or national-scale estimates for belowground biomass or deadwood pools are currently available for comparison. Pool definitions for soil inventories are different from our pools which are based on IPCC definitions (IPCC 2003). The closest comparison is between the inventory-based estimate for unfrozen mineral soil that would include mineral soil C to 100 cm depth plus C in the associated

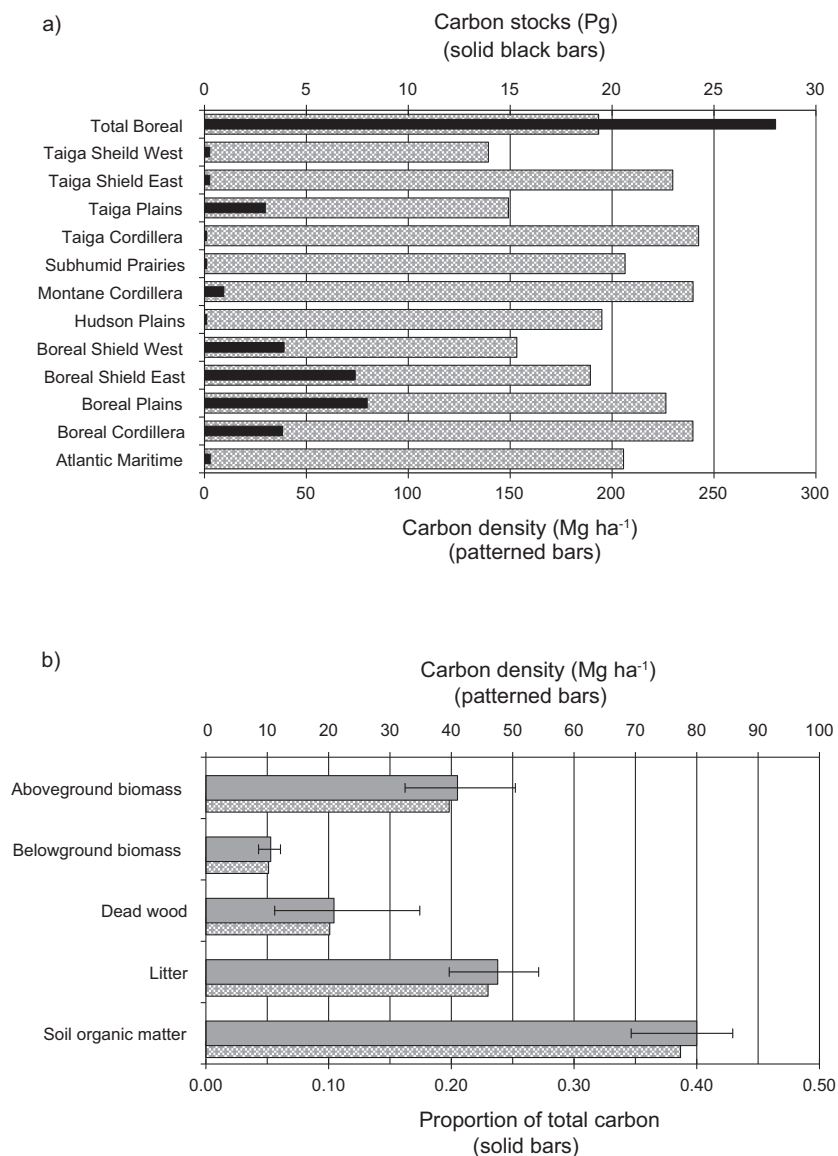
soil organic horizons (Table 1;  $138 \text{ Mg}\cdot\text{ha}^{-1}$ ) and our estimate for L (litter) + SOM (mineral soil) ( $123 \text{ Mg}\cdot\text{ha}^{-1}$ ) in Fig. 5b.

We were able to estimate proportions and C densities for C pools based on published data for the middle and southern taiga regions of the Russian boreal forest (Shvidenko and Nilsson 2003). Carbon stocks (proportion; density in  $\text{Mg}\cdot\text{ha}^{-1}$ ) were similar to our values for the aboveground biomass ( $0.18$ ;  $34.8$ ) and belowground biomass ( $0.04$ ;  $9.2$ ) pools but were lower for the deadwood ( $0.05$ ;  $10.2$ ) and litter ( $0.07$ ;  $13.3$ ) pools and higher for the soil organic matter pool ( $0.66$ ;  $131.8$ ). Carbon densities for aboveground biomass and soil organic matter pools reported for boreal forests in Finland ( $34$  and  $72 \text{ Mg}\cdot\text{ha}^{-1}$ , respectively; Kauppi et al. 1997) are similar to our estimates.

The NEE (in units of C) with the atmosphere was estimated at  $-28 \pm 16 \text{ Tg year}^{-1}$  during 1990–2008 (Fig. 6), where the negative sign denotes net removal of C from the atmosphere and the range indicates standard deviation of the time series, not overall uncertainty. An amount equivalent to more than half of the C removed from the atmosphere was transferred out of the forest by harvesting ( $17 \pm 3 \text{ Tg year}^{-1}$ ) with the balance accumulating in living biomass ( $7 \pm 25 \text{ Tg year}^{-1}$ ) and DOM pools ( $4 \pm 10 \text{ Tg year}^{-1}$ ). More C was transferred out of the managed boreal forest by wildfires



**Fig. 5.** (a) Carbon density (patterned bars) and total ecosystem C stocks (solid black bars) for the entire managed boreal forest area and by ecozone; and (b) C density (patterned bars) and proportion of total ecosystem C (solid bars) for the five C pools defined by the IPCC. In (a), three ecozones appear to have zero stocks because their area in the managed forest is very small. In (b), error bars indicate range of values across ecozones and the IPCC litter pool includes foliar litter and fine woody debris. The IPCC pools for the CBM-CFS3 are defined as follows: aboveground biomass, all live tree biomass above the soil including stem, stump, branches, bark and foliage; belowground biomass, all live tree roots; dead wood, all nonliving woody organic matter standing or lying on the ground ( $\geq 75$  mm diameter) not contained in the litter; litter, all nonliving organic matter with a diameter  $< 75$  mm including organic soil horizons above the mineral soil; soil organic matter, organic carbon in the mineral soil to 100 cm depth.



than harvesting. Fire and insect disturbances also cause large fluxes of C within the ecosystem (Fig. 6). High interannual variability in the areas affected by fires and insect outbreaks cause high interannual variability in Canada's boreal forest C budget (Kurz and Apps 1999; Chen et al. 2000; Stinson et al. 2011). Carbon impacts from insect outbreak activity (aspen defoliators, spruce beetle, and eastern hemlock looper) in boreal forests during 1990–2008 were relatively small but have been high in the past (Kurz and Apps 1999) and are anticipated to be high again in the future (Dymond et al. 2010). In accordance with international reporting guidelines (IPCC 2003), reported estimates of C emissions and removals refer to the area “forest land remaining forest land” and do not include fluxes associated with deforestation, which are

reported in the land-use category that follows deforestation, e.g., “forest land converted to cropland” (see section 3.2.3).

The largest annual C fluxes in boreal forests are the uptake of C by NPP and the release by  $R_h$ . Most of the C uptake through NPP is balanced by C release through  $R_h$  (Fig. 7). Approximately one tenth of the C taken up by NPP has accumulated in the ecosystem as NEP ( $NEP = NPP - R_h$ ). Four fifths of that C were transferred out of the ecosystem, either directly into the atmosphere by fires or transferred to the wood product sector by harvesting. During the period 1990–2008, only 2.5% of the incoming C remained in the system (NBP) after accounting for all losses.

Interannual variations in NPP and  $R_h$  are not fully captured in the estimates presented in Fig. 6 because the impacts of environ-

**Table 1.** Distribution of soil carbon (C) stocks in Canada's boreal zone.

Soil type	Area ( $\times 10^3$ km <sup>2</sup> )	C stock (Pg C)	C density (Mg·ha <sup>-1</sup> )
<b>Peatlands</b>			
Frozen	392 (341)	41 (34)	1046
Unfrozen	658 (290)	96 (36)	1459
Total	1050 (631)	137 (70)	130
<b>Mineral soils</b>			
Frozen	614	33	537
Unfrozen	2752	38	138
Total	3366	71	210

**Note:** Estimates based on data available in Tarnocai (2006); Bhatti and Tarnocai (2009), and Carlson et al. (2010). Values in parentheses are the areas considered sensitive to climate change by Tarnocai (2006). The definition of pools in the data used in these estimates differs from the IPCC definition of pools (IPCC 2003).

mental variation and trends remain poorly understood as discussed in section 3.3.1. Environmentally driven variations in growth and mortality rates are not represented in the NFCMARS nor are interannual variations in  $R_h$  resulting from climate variability because simulation runs conducted for reporting purposes use constant average climate conditions.

The NPP for Canada's managed boreal forests, estimated at  $438 \pm 2$  Tg C year<sup>-1</sup> with a flux density of 302 g C m<sup>-2</sup> year<sup>-1</sup> (Figs. 6 and 7), is similar to the estimates reported by Chen et al. (2000); Li et al. (2003); Zheng et al. (2003), and Kang et al. (2006) but lower than the estimates reported by Gower et al. (2001) or Kimball et al. (2006). The low C sequestration ratio (NBP:NPP = 0.03) calculated for Canada's managed boreal forests is attributed to the relatively old age of these forests and the high natural disturbance rates during the period 1990–2008.

Contemporary boreal forest C dynamics are driven by current forcing mechanisms and the legacy effects of historical environmental conditions and past disturbance events. Canada's boreal forests are old on average (85 years in 2008, Stinson et al. 2011) compared with forests in Europe (48 years, Böttcher et al. 2008) and the conterminous US (Pan et al. 2011b). As a result, their C density is high relative to the theoretical C-carrying capacity, defined as the C stock that would be sustained on the landscape under the current set of environmental conditions and disturbance regimes (Fig. 8). Having high C density relative to the theoretical C-carrying capacity suggests limited capacity to take up additional C unless climate change brings about an increase in the carrying capacity or unless unnaturally high C stocks are maintained through management (Lemprière et al. 2013).

In a recent global study of forest C dynamics (Pan et al. 2011a), all of Canada's managed forests (230 Mha) were entirely included in the global boreal stratum, and a reduction in biomass C sink by half between the 1990–1999 and 2000–2007 estimates was reported. This decline in Canada's C sink is strongly affected by the impacts of the mountain pine beetle outbreak (Kurz et al. 2008c) that occurred outside the boreal forest and, therefore, does not affect the results shown here.

### 3.2.2. Unmanaged forest area in the boreal zone

Between the managed boreal forest to the south and the Arctic tundra to the north are some 125 Mha of unmanaged boreal forest, plus 39 Mha of low productivity "other wooded land" with canopy closure or tree heights below the thresholds of forest definitions (Fig. 3).

Traditional timber inventories do not include information about the unmanaged forest, although older national forest inventories (Bonnor 1985) included information about this area, which served as input to early analyses of Canada's forest C budget (Kurz et al. 1992; Kurz and Apps 1999). Canada's NFCMARS relies on more recent and more detailed forest inventory infor-

mation and was designed to meet international GHG reporting requirements, which are limited to areas subject to forest management (IPCC 2003). Canada monitors and reports deforestation activities in all forest areas (see section 3.2.3), including the unmanaged boreal forest. Canada's new NFI (Gillis et al. 2005) covers the entire boreal forest area with 2 km  $\times$  2 km photo plots located on a 20 km  $\times$  20 km grid and groundplots on a subset of these plots, and future information on the C budget of the unmanaged forest area will be informed by these NFI data. Additional efforts to quantify forest biomass C stocks are under way using remote sensing, radar, and airborne LiDAR technologies (Boudreau et al. 2008; Magnussen and Wulder 2012).

The C balance of the unmanaged boreal forest is largely determined by natural processes affecting growth, mortality, and decomposition. Fires, ignited by lightning strikes, are the dominant natural disturbance (Fig. 4), as large-scale insect outbreaks play a secondary role. By definition, there is no commercial forest harvesting in the unmanaged boreal forest.

Forest productivity in the unmanaged boreal forest is very low (Bickerstaff et al. 1981), limited by cold temperatures, permafrost, short growing seasons, low decomposition rates, and nutrient availability (Maynard et al., *In press*; Lavigne et al., *Manuscript in preparation*). Although biomass C stocks are low, DOM and soil C pools can be very large, often associated with deep organic soils, peatlands, and permafrost (Schoor et al. 2009; Tarnocai et al. 2009; see also section 3.3.3).

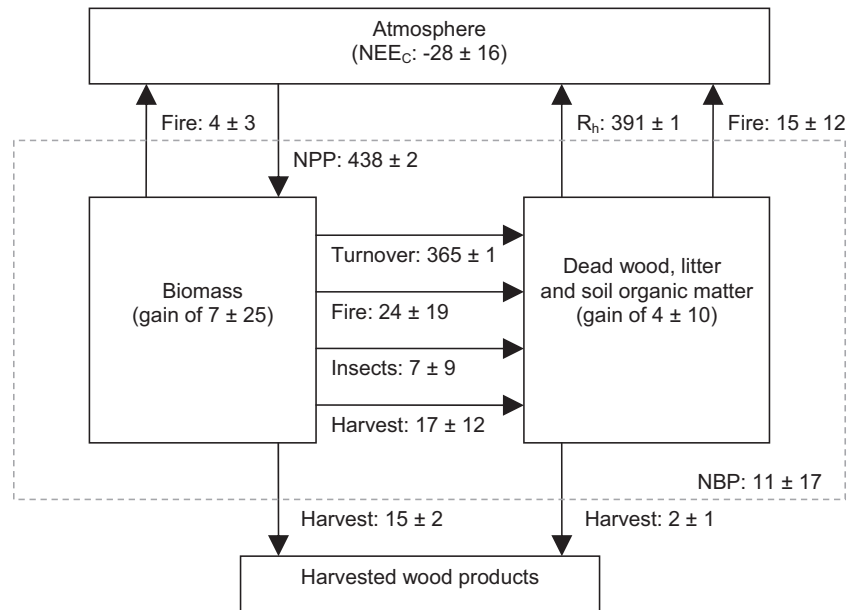
In theory, it should be possible to estimate the C balance of the unmanaged boreal forest from inversion models. However, current estimates derived from inversion models do not have the spatial resolution to permit the extraction of results for just the unmanaged forest area because they cover much larger geographic areas and, within these, they estimate the contribution from all terrestrial and aquatic surface fluxes. Moreover, in several recent comparisons inversion models tended to estimate substantially larger sinks than inventory-based approaches (Hayes et al. 2012; Huntzinger et al. 2012); and for the North American boreal zone, inversion-based model estimates do not agree on the sign of the net flux (Hayes et al. 2011). Over the past 50 years, the amplitude of the seasonal CO<sub>2</sub> exchange in the circumpolar boreal zone has increased by about 67% (Graven et al. 2013). Atmospheric CO<sub>2</sub> measurements indicate that both growing season uptake and dormant season release have increased, but it is not yet possible to determine the net change in these two fluxes which need not be in balance (Graven et al. 2013).

Similarly, the few available and as yet poorly constrained estimates of North American C fluxes derived from process models do not agree on the magnitude and, in some cases, the direction of the net C fluxes (Hayes et al. 2011, 2012; Huntzinger et al. 2012). Many process models include assumptions about productivity enhancements due to atmospheric CO<sub>2</sub> concentration increases, warmer temperatures, and longer growing seasons (Chen et al. 2000; Balshi et al. 2007; Hayes et al. 2011). However, these modelling assumptions are still poorly constrained (see section 3.3; Girardin et al. 2011). Process models are particularly sensitive to the assumptions about permafrost dynamics in the northern boreal forests. Increasing the complexity of representation of the temporal dynamics of the active layer in the terrestrial ecosystem model (TEM) greatly affected the conclusions about the magnitude of the increase in emissions from heterotrophic respiration with temperature increases (Hayes et al. 2011). Given these large uncertainties in estimates of net C fluxes, we do not report estimates of C fluxes for Canada's unmanaged boreal forest.

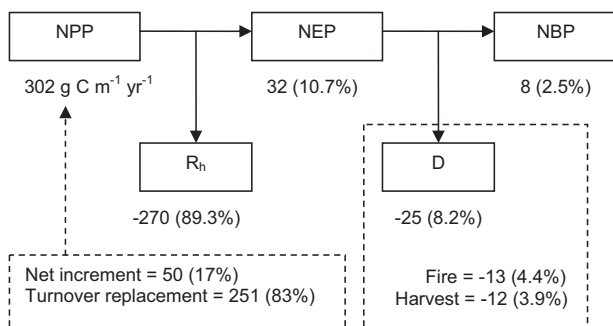
### 3.2.3. Land-use change: deforestation and afforestation

For the purposes of international reporting, deforestation is defined as "direct human-induced conversion of forested land to nonforested land" and afforestation is defined as "the direct

**Fig. 6.** Carbon budget of Canada's managed boreal forests during 1990–2008 based on extraction of results from the NFCMARS as described in *Stinson et al. (2011)*. Means and standard deviations are for the 19 annual estimates and rounded to the nearest Tg C year<sup>-1</sup>. The net exchange of C between these forests and the atmosphere resulted in a net removal of C from the atmosphere (negative net ecosystem exchange of C; NEE<sub>C</sub> in units of C), a net accumulation in the forest (positive net biome production; NBP), and a transfer of C into harvested wood products. The release over time of C from harvested wood products is an additional transfer to the atmosphere, not quantified here. NPP, net primary production.



**Fig. 7.** Estimated ecosystem C fluxes during 1990–2008 based on extraction of results from the NFCMARS as described in *Stinson et al. (2011)* for managed forests of the boreal zone. All fluxes are landscape averages ( $\text{g C m}^{-2} \text{ year}^{-1}$ ) over the 19-year time period. Heterotrophic respiration ( $R_h$ ), net ecosystem production (NEP), disturbance transfers out of the ecosystem (D) and net biome production (NBP) are also expressed as percentages of net primary production (NPP). NPP was estimated in the CBM-CFS3 as the sum of net biomass increment and replacement of above- and below-ground biomass turnover. D is broken down into direct emissions to the atmosphere by fire and wood harvest transfers out of the ecosystem.



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" ([http://unfccc.int/files/meetings/workshops/other\\_meetings/application/pdf/11cp7.pdf](http://unfccc.int/files/meetings/workshops/other_meetings/application/pdf/11cp7.pdf), p. 58; IPCC 2003). Reforestation is defined like afforestation but occurs on land that was not forested in 1989 but may have been forested at some time during the 50 years prior to reforestation. Historical land-use change can alter forest ecosystem structure and its legacy can influence the contemporary C balance. However, the historic rates of deforestation and afforestation in Canada's boreal zone are low, very

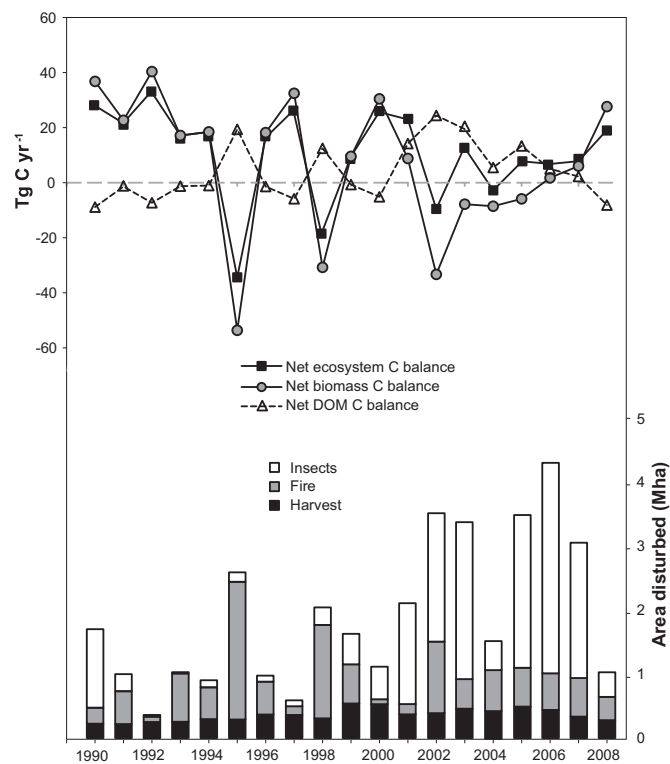
different from the situation in Scandinavia and Russia, and comparable to the situation in Alaska. In Scandinavia, boreal forests are recovering from extensive conversion and exploitation in the 19th and 20th centuries and have since been a C sink owing to increases in forest area and in C density (*Kauppi et al. 2009, 2010*). In Russia, large tracts of boreal forest were converted to agricultural land during the era of the Soviet Union; and after its collapse and the reduction in agricultural land use, some 30–40 Mha of land have returned to forest (*Baumann et al. 2011; Pan et al. 2011a; Quegan et al. 2011*) and are contributing to the observed C sink in Russia's forests. The boreal forests of Alaska, like those of Canada, have not been subjected to extensive land-use change, and their C dynamics are largely driven by natural disturbances (*McGuire et al. 2006*). Globally, estimates for contemporary C budgets indicate that deforestation contributes about 12%–15% (*van der Werf et al. 2009; Pan et al. 2011a*) of the anthropogenic C emissions to the atmosphere; but in Canada's boreal forest, deforestation contributed less than 2% of Canada's national CO<sub>2</sub> emissions in 2010 (*Environment Canada 2012*).

We briefly describe historic land-use change information and provide an assessment of the impact of contemporary land-use change involving forests in Canada's boreal zone following the definitions and conventions of the United Nations Framework Convention on Climate Change (UNFCCC). Technical details of methodologies and approaches are provided in *Appendix A*.

### 3.2.3.1. Historical land-use change

Past land-use change has overall been small in Canada's boreal forest, with some regional exceptions. Agricultural development has been important along the southern fringe of the boreal in the Prairie provinces (Alberta, Saskatchewan, and Manitoba) (*Hobson et al. 2002; Fitzsimmons et al. 2004*), the Peace River region of Alberta and British Columbia, and locally in the southern boreal Clay Belt region (*Vanderhill 1988*) of western Quebec and eastern Ontario. As European settlers moved west, forest was converted to agricultural land (*Hobson et al. 2002*). In recent decades, abandon-

**Fig. 8.** Estimated net C fluxes from the managed forest in Canada's boreal zone, including biomass, dead organic matter (DOM, including dead wood, litter, and soil organic matter), and ecosystem total (net ecosystem C balance across the boreal zone). Positive stock changes indicate increase (forest sink) and negative stock change indicate loss (source). Fire and most harvests are stand-replacing. Insect impacts range from growth reductions with no mortality to 70% stand mortality. In accordance with international reporting guidelines, neither the storage of harvested C in harvested wood products (section 3.2.4) nor the emissions from deforestation (section 3.2.3) are included.



ment of agricultural land has occurred in regions of eastern Canada (Kent 1966; Vanderhill 1988; Foote and Grogan 2010).

Analysis of the total area of agricultural land in the boreal forest of the Prairie and Peace River regions indicates that 9.1 Mha is currently in agricultural use (T. Fiset, personal communication, 2011). The proportion of this land that was forested prior to European agriculture development is unknown; however, assuming 80%–90% was forested suggests some 7.7 Mha of deforestation owing to agricultural use over the last 120 years. Several data sources were used (see Appendix A) to estimate the combined total land-use change throughout Canada's boreal zone from agriculture, hydroelectric reservoirs or settlements (settlement includes towns, main roads, mines, and other infrastructure) at 11.8 Mha or approximately 2% of the boreal zone. Approximately 80% of this is agriculture land and 13% flooded reservoir land.

### 3.2.3.2. Contemporary land-use change

To meet international reporting obligations for GHG emissions and removals and to inform policy makers, the public, and land managers, Canada has implemented a deforestation monitoring program that detects, quantifies, and reports deforestation activities in Canada's entire forest land area (Leckie et al. 2002, 2006; Environment Canada 2009). The international guidelines for accounting of GHG emissions and removals make a clear distinction between changes in land cover and changes in land use. Forest clearing and subsequent nonforest land use such as agriculture,

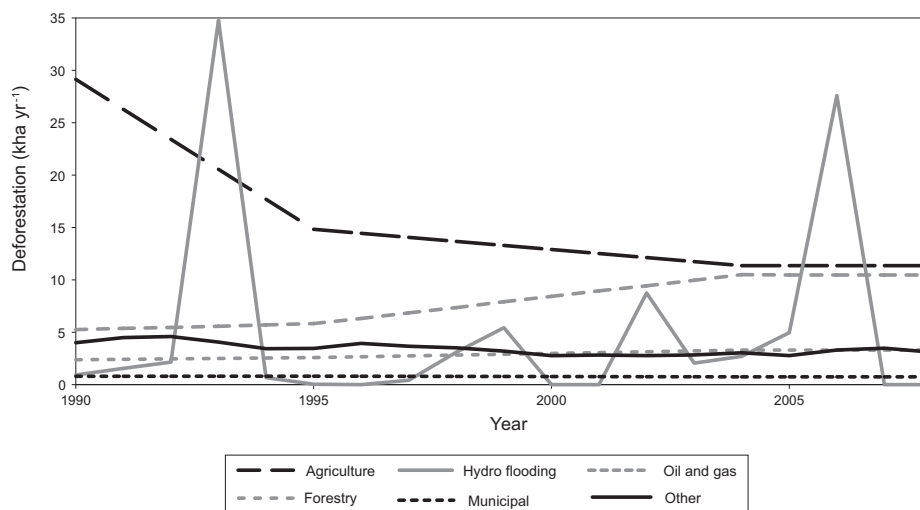
settlements, or road infrastructure are considered deforestation. Under UNFCCC rules, the emissions associated with this change in land use are accounted for in the new land-use sector responsible for the deforestation. The rule that the final land-use category is accountable for all emissions associated with a land-use change also applies in the case of afforestation. Thus, emissions associated with brush clearing, site preparation of nonforest land prior to planting, and conversion back to forest uses are accounted for in the forest category, as are all subsequent sinks in the afforested land area. Forests that are logged and subsequently re-establish through natural regeneration or planting remain in the land-use category forest, even if for some period of time tree cover is absent. Emissions associated with harvesting, slash burning, and decomposition are reported in the forest category. This convention poses some challenges for the acquisition of land-cover information through remote sensing for three reasons. First, recent reports of gross forest-cover loss (Hansen et al. 2010) are based on observed forest-cover changes owing to anthropogenic or natural causes but are not a measure of loss of forest area or deforestation (Kurz 2010; Reams et al. 2010; Wernick et al. 2010). Second, observed forest-cover loss does not imply deforestation events because these also require the transition to a nonforest land use. Third, areas that are temporarily devoid of forest cover can still contribute emissions that need to be reported in the managed forest's GHG balance.

Quantification of the area annually deforested in Canada is based on interpretation of a time series of Landsat satellite images for three time periods (ca. 1975–1990, 1990–2000, and 2000–2008). Mapping is conducted on a network of sampling cells and supported by ancillary data, high-resolution images, and sometimes verification through aerial observation or ground visits (Leckie et al. 2006, 2009). Area deforested by year, sector, and forest type is then used in the CBM-CFS3 (Kurz et al. 2009) to estimate the associated direct emissions in the year of deforestation from clearing and, where appropriate slash burning, and the delayed post-deforestation emissions from the decay of residual DOM such as stumps, slash, and litter.

Between 1990 and 2008, the area annually deforested in the boreal zone was 36 kha year<sup>-1</sup> or about 0.016% of the forested area in the boreal zone (Table A1). Annual deforestation rates have declined from 42 kha year<sup>-1</sup> around 1990 to about 30 kha year<sup>-1</sup> from 1995 onward (Fig. 9; Table A1). Spikes in deforestation rates are the result of hydroelectric developments and the flooding of forest by reservoirs. Cumulative deforestation was 674 kha over the period 1990–2008 or about 0.3% of the forested area of the boreal zone. Average direct and residual annual emissions due to deforestation (excluding residual emissions from pre-1990 deforestation events) were 1.6 Tg C year<sup>-1</sup> (6.2 Tg CO<sub>2</sub>e year<sup>-1</sup>; CO<sub>2</sub>e, carbon dioxide equivalent) or cumulatively 30.7 Tg C (118 Tg CO<sub>2</sub>e) over the 19-year period. In addition, 0.72 Tg C year<sup>-1</sup> (2.6 Mt CO<sub>2</sub>e year<sup>-1</sup>) have been transferred to the forest product sector annually, or cumulatively 13.7 Tg C (50.0 Tg CO<sub>2</sub>e) over the 19-year period. These transfers have been reported as immediate emissions to the atmosphere.

The magnitude and causes of deforestation differ by region. In the eastern boreal forest (Ontario, Quebec, and Newfoundland), forest roads are the most important cause of deforestation, averaging 1.9 kha year<sup>-1</sup> over the period 1990–2008 or 42% of the area deforested (excluding hydroelectric reservoir developments). In the western boreal forest, forest roads contribute 0.9 kha year<sup>-1</sup> to deforestation. In the southern boreal forest, deforestation due to forest roads is ongoing because of increasing road density and expansion of harvest into previously unaccessed regions, whereas in the Great Lakes St. Lawrence forest region and Atlantic Canada the main and secondary access roads to the forest are largely established. Thus, as the infrastructure of forest roads matures, deforestation from forest roads is likely to decrease.

Fig. 9. Area annually deforested in the boreal zone of Canada (kha year<sup>-1</sup>) during 1990–2008 by major causal categories.



Another significant source of deforestation in the eastern boreal forest is hydroelectric power development in Quebec where several very large hydroelectric reservoirs, dams, and associated transmission lines resulted in about 489 kha of deforestation in the late 1970s and in the 1980s. In the period 1990–2008, four main hydroelectric developments contributed spikes in the deforestation rates (Fig. 9; Table A1).

In the Prairies, agriculture development along the southern fringe of the boreal zone is the largest contributor to deforestation, although land-use allocations such as forest management agreements and zoning for forest and agricultural land use limit expansion of agriculture into forests. Agriculture deforestation rates in the Prairies were about 48 kha year<sup>-1</sup> in the 1970s and early 1980s. In the period 1990–2008, agriculture-related deforestation declined from 28 to 10 kha year<sup>-1</sup>.

Oil and gas infrastructure development is an important contributor to deforestation in Canada (Table A1), most of it in the boreal forest of Alberta and, to a lesser extent, northeast British Columbia, the southeast corner of the boreal zone in Saskatchewan and the Northwest Territories. Oil and gas infrastructure development is one of the few industrial categories of deforestation that has increased over time, although with temporal fluctuations affected by economic conditions and prices. Deforestation associated with surface oil sands projects has increased recently. Emissions from fossil sources are reported in the National GHG Inventory (Environment Canada 2011).

The boreal zone is characterized by very low population density and few large communities. Urban growth and rural residential development in the boreal zone contribute only 2%–3% of the area deforested from 1990 to 2008.

Conversion of nonforest land back to forest has been very limited in Canada's boreal zone when compared with the boreal zones in Russia and Scandinavia because of the relatively short history of land-use change and the small proportion of the boreal forest area that has been converted to agricultural land uses (White and Kurz 2005). Statistics compiled for Canada's national GHG inventory reporting indicate that conversion of nonforest to forest land (afforestation and reforestation in the UNFCCC definition) in the boreal zone was limited to about 1 kha year<sup>-1</sup> or about 35% of the reported afforested area for Canada during the period 2000–2008 (data derived from NFCMARS as described in Stinson et al. (2011) and Environment Canada (2011)). Abandonment of agricultural land and subsequent regrowth of forest will contribute to further increases in forest area, but no national-scale statistics are available at present and the area involved, while potentially larger than the reported area afforested, is still small.

The annual C sink contribution of the reported small area of afforested land in Canada's boreal zone is estimated at less than 1 Mt CO<sub>2</sub>e (Environment Canada 2012), insignificant at current rates of afforestation and well within the uncertainty of the C balance of the managed boreal forest zone. Afforestation offers opportunities to increase C sequestration in forest areas (Lemprière et al. 2013) but at considerable cost. Moreover, some of the climate mitigation benefits from afforestation may be offset from the associated changes in albedo (Bernier et al. 2011).

#### 3.2.4. Harvested wood products

The annual transfer of C from Canada's managed boreal forest to the forest product sector was 17 Mt C year<sup>-1</sup> (Fig. 6) over the period 1990–2008. The current default accounting guidelines of the UNFCCC assume that C removed from the forest replaces C in harvested wood products (HWPs) derived from harvest in prior years such that the total pool of HWPs remains constant. The additions to the HWP pool are assumed equal to the releases from the pool, and the simplifying accounting assumption is that all C added to the HWP pool is immediately emitted to the atmosphere (IPCC 2003). In reality, however, the global HWP C pool has not yet reached steady state and is still increasing in size, storing additional C in products in use (houses, furniture, etc.) and in landfills (Micales and Skog 1997; Apps et al. 1999; Chen et al. 2008). The simplifying accounting assumption does not consider where and when emissions actually occur, nor does it provide incentives for countries to consider climate change mitigation through the management of HWP (Lemprière et al. 2013).

Harvested wood products store C for years to decades depending on the end use; and even after disposal in landfills, a large proportion of the C is retained (Micales and Skog 1997; Apps et al. 1999). Of the 323 Mt C cumulative harvest from Canada's managed boreal forest, we estimate that 56.9% (or 184 Mt C) have not yet been released to the atmosphere. Some of this C has merely replaced C released from HWPs that were harvested prior to 1990, and the remainder contributed to net increases in the HWP and landfill C pools.

The use of HWP has also contributed to reductions in emissions in other sectors, through the substitution of emissions-intensive products such as steel, concrete, and plastics with low emissions, renewable HWP (Nabuurs et al. 2007; Hennigar et al. 2008; Sathre and O'Connor 2010; Werner et al. 2010). Forest-sector mitigation options using HWP from Canada's boreal forests are discussed by Lemprière et al. (2013).

### 3.3. Processes not included in the contemporary C estimates

The Canadian NFCMARS and its core model, the CBM-CFS3, are currently used for national reporting, policy analyses, and scientific research. This system has also provided the most complete C balance estimates for the managed forest area within the boreal forest zone of Canada. Hence, the detailed, quantitative C balance presented in this review (section 3.2.2) has concentrated on estimates from NFCMARS. However, some processes that can affect C fluxes are not currently represented in the system, primarily because they are poorly understood, lacking data for their quantification, or not easily represented spatially across the boreal zone. In this section, we identify and discuss these recognized deficiencies that should be taken into consideration along with the estimates provided in the previous section.

#### 3.3.1. Response to environmental changes

The general patterns of stand-level C dynamics are well understood, although large uncertainties and knowledge gaps remain regarding the quantification of fluxes and the factors controlling site-specific and interannual responses to changing environments and anthropogenic and natural disturbances. Eddy covariance studies show significant interannual variability in the stand-level C balance of boreal forests (Griffis et al. 2003; Amiro et al. 2006; Kljun et al. 2006; Barr et al. 2007; Dunn et al. 2007; Grant et al. 2009; Goulden et al. 2011; Coursolle et al. 2012). Spatially synchronous interannual variability in NEE due to environmental variability over large areas and of the magnitude observed in some EC studies have a substantial effect on annual estimates of the landscape-level ecosystem C balance. This is reflected in part in the large interannual variability in global estimates of the net contribution of terrestrial ecosystems to the global C budget (Le Quéré et al. 2009).

Changes in NPP and  $R_h$  may be caused by many changes in environmental factors including the ongoing increase in  $\text{CO}_2$  concentration, atmospheric N deposition, and changes in temperature and precipitation regimes. While temperature effects on  $R_h$  are represented in the CBM-CFS3, the effects of other environmental changes on decomposition (see section 4.2) are not currently included (e.g., Smyth et al. 2011). Yield tables based on sample plot measurements used in NFCMARS account for the impacts of environmental change up to the time that plot measurements were taken. However, under conditions of progressive changes in environmental conditions including atmospheric  $\text{CO}_2$  increases, yield tables derived from compilations of plot measurements taken over long periods of time introduce systematic bias and only partly reflect the impacts of environmental change (Hember et al. 2012). More recent and ongoing impacts from changing environmental conditions and variability (e.g., Boisvenue and Running 2006) are currently not explicitly accounted for in Canada's NFCMARS.

The global atmospheric  $\text{CO}_2$  concentration is now nearly 400 ppm, well above preindustrial levels of 278 ppm (Tans and Keeling 2011) and continues to increase at an accelerating rate (Peters et al. 2012). A doubling of atmospheric  $\text{CO}_2$  concentration has been shown to increase productivity, in some cases up to 23% in temperate forest experiments (Norby et al. 2005). The effect of  $\text{CO}_2$ , however, is currently thought to be site-specific (Nowak et al. 2004), constrained by available N and nutrient levels (Lloyd 1999; Beedlow et al. 2004; Norby et al. 2010), with  $\text{CO}_2$  levels contributing to higher productivity on specific sites for specific species (e.g., Cole et al. 2010; Hember et al. 2012). It is not known, however, whether these growth enhancements will continue as atmospheric  $\text{CO}_2$  concentration increases. There is likely a smaller effect in boreal forests than that observed in the temperate experiments because of the colder temperatures, which are estimated to limit the increase in NPP to less than 16% (Hickler et al. 2008; Pinsonneault et al. 2011). However, tree-ring analysis of

growth trends under natural conditions in the Boreal Plains Ecozone of Manitoba has so far been unable to detect a  $\text{CO}_2$  fertilization signal in the observed historic trend (Girardin et al. 2011). Possible methodological biases make it difficult to interpret reports of tree-ring based detection of forest productivity trends (Brienen et al. 2012). Increased  $\text{CO}_2$  concentrations can also affect photosynthate allocation to roots and decomposition rates, as discussed in section 4.2.

Increases in temperature may lead to an increase in productivity (Fitter and Hay 2002). A larger temperature effect has been found in high-latitude, cold-limited systems where tree-ring chronologies show good agreement with century timescale warming trends (Briffa et al. 2008). This effect could be due to the expected physiological response or an increase in growing season length (Richardson et al. 2010b) as reported by Kimball et al. (2004) for the boreal regions. The positive effects of increased temperatures can be limited by temperature-induced drought stress, where reduced water availability increases tree mortality (Michaelian et al. 2011; Peng et al. 2011) and limits forest productivity and net biomass increment in some interior regions of the boreal forest (Hogg et al. 2008; Beck et al. 2011; Ma et al. 2012). The analyses of Silva et al. (2010) show an increase in water use efficiency under elevated  $\text{CO}_2$  on Canadian boreal sites, but a decline in growth partially attributed to the warming-induced stress and partially attributed to additional nonidentified stressors. Temperature changes affect tree populations at the northern and southern edges of the boreal forest in different ways (Silim et al. 2010).

Nitrogen (N) is the primary limiting nutrient in Canadian boreal forest ecosystems (Tamm 1991; Maynard et al., *In press*), in spite of increases in atmospheric N deposition associated with anthropogenic pollution (LeBauer and Treseder 2008). Globally, atmospheric deposition originating from anthropogenic sources has now become the dominant source of N and varies from  $\sim 0$  to  $60 \text{ kg ha}^{-1} \text{ year}^{-1}$  (LeBauer and Treseder 2008). In the boreal regions, N deposition remains relatively low (mostly below  $3 \text{ kg ha}^{-1} \text{ year}^{-1}$ ) compared with temperate forest ecosystems (Bobbink et al. 2011; Gundale et al. 2011). In addition, a recent study has shown that bryophytes likely limit woody plant acquisition of ambient anthropogenic N inputs, especially at low deposition rates that are commonly experienced in the boreal zone of Canada (Gundale et al. 2011). Growth enhancement from recent N deposition is, therefore, likely to be nonexistent or extremely localized in the boreal forests of Canada.

The factors discussed in this section are currently not included in Canada's estimates of boreal forest carbon budget but are likely to have affected the interannual variability, regional distribution, and trends of C sources and sinks. However, uncertainties about the response of ecosystem C fluxes to changes in environmental drivers remain high, with existing process models not agreeing on the magnitude and sometimes the direction of the net changes in fluxes (Wang et al. 2011; Huntzinger et al. 2012). Further quantitative analyses of the impacts of environmental drivers on stand growth and mortality (Hember et al. 2012) and on ecosystem C fluxes and ongoing monitoring of forest dynamics are underway to reduce these uncertainties.

#### 3.3.2. Disturbances other than fire and insects

Only major disturbances from fire, harvesting, and large insect outbreaks (e.g., mountain pine beetle, aspen defoliators, eastern hemlock looper, and spruce beetle) are included in the current C balance estimates applicable to the boreal zone. Some additional disturbances by insects, pathogens, and windthrow are likely to further reduce C sink strength in specific regions of the boreal zone but are not well quantified.

The CBM-CFS3 and other models based on empirical yield tables assume that the impacts of diseases endemic to a region are included in yield tables derived from measurements in permanent

or temporary sample plots. The impacts of new diseases or disease epidemics will only be reflected in regional C budgets if these are represented as disturbance events that modify productivity, mortality, or transfers to DOM or HWP (via salvage logging). For example, the recent widespread aspen decline and dieback in the southern boreal forests of Alberta and Saskatchewan (Hogg et al. 2008; Michaelian et al. 2011) are currently not accounted for in Canada's estimates of GHG emissions. Work is in progress to quantify both the growth reductions and increased mortality associated with widespread drought and defoliators.

Sturrock et al. (2011) reviewed the current understanding of climate change impacts on pathogens and diseases that occur at endemic and epidemic levels. *Dothistroma* needle blight, lodgepole pine dwarf mistletoe, several root diseases, and other pathogens are predicted to respond to climate change; but with complex disease host interactions, the impacts are difficult to forecast. Aspen dieback or decline in Alberta and Saskatchewan in the southern boreal zone could possibly be the beginning of widespread species dieback in the boreal zone attributed to changing weather and climate patterns and interactions with defoliating insects (Hogg et al. 2002, 2008; Michaelian et al. 2011). While these impacts of climate change will be difficult to diagnose and quantify, it is generally accepted that the risk of large-scale disease outbreaks will increase with increased climate variability partly because pathogens and disease can adapt to new climate conditions faster than tree species (Sturrock et al. 2011). Responses of the boreal forest to insects, pathogens, and drought under a changing climate are discussed in further detail by Price et al. (2013).

Windthrow in the Canadian boreal forest is currently not explicitly modelled in the NFCMARS. Windthrow has been reported as one of the most important factors driving succession in Russian boreal forests (Ulanova 2000) and extreme windthrow events in Scandinavian boreal forests have been shown to affect C budgets (Lindroth et al. 2009). Windthrow impacts in Canada's boreal forest appear to be limited to occasional local events (Flesch and Wilson 1999; Bouchard et al. 2009) and are a minor driver compared with other natural disturbances. However, uncertainties remain because windthrow extent is typically not monitored or reported by provincial resource management agencies. Windthrow damaged ~600 000 m<sup>3</sup> of timber in one storm in two regions of the Quebec boreal forest (Ruel 2000) and at least 25% of canopy tree mortality in the Clay Belt of the Ontario and Quebec boreal forest was attributed to windthrow by the time stands reached 200 years of age (Harper et al. 2003). While the extent of wind damage throughout Canada's boreal forest is not quantified, assumptions in the NFCMARS are that background levels of wind-induced mortality are reflected in the existing yield tables and only catastrophic windthrow events should be represented as additional disturbance events when they occur.

The explicit C accounting of additional disturbance types would require the monitoring of areas affected by these disturbance types, as well as their effects on growth, mortality, and transfers to DOM or HWP following salvage logging. However, based on the limited data available, the magnitude of this reduction is currently expected to be well within the overall uncertainty of existing C balance estimates. But if the frequency or intensity of these disturbances increases with climate change, then their impacts on C balances could become much more significant and they should then be included in NFCMARS.

### 3.3.3. Permafrost (including peatlands)

Permafrost is defined as ground that remains at or below 0 °C for two or more consecutive years, which may or may not contain ice in addition to any combination of mineral soil, organic matter, and bedrock (e.g., Brown and Pewe 1973; Grosse et al. 2011). Here we use the definition of mineral soil provided in the Canadian

glossary of soil terminology (<http://sis.agr.gc.ca/cansis/glossary/m/index.html>) as soils consisting predominantly of mineral matter including an organic layer up to 40 cm thickness (mixed peat) or 60 cm thickness (fibric peat). Mineral soils are distinguished from organic soils that are classified in the Organic order or the Organic great group of the Cryosolic order (Soil Classification Working Group 1998) that in the boreal zone are largely composed of moss-derived peat.

A large area (~647 Mha) of Canada is affected by permafrost and 70% of this area (388 Mha) occurs in the boreal zone (Fig. 3). In the continuous permafrost zone, permafrost occurs everywhere beneath the ground surface except below large bodies of water, whereas it underlies varying proportions of the land area in the discontinuous permafrost zone (Hegginbottom et al. 1995). Aboveground productivity in these zones tends to be low, but decomposition rates are even lower so that larger amounts of ecosystem C are stored in mineral soils and organic soils of peatlands (Wieder et al. 2006; Bhatti and Tarnocai 2009) than in aboveground biomass. For example, Vitt et al. (2000) estimated that peatlands, which dominate the discontinuous permafrost zone, in continental western Canada contain 42 Pg C stored as peat and 6 Pg C in the aboveground biomass.

#### 3.3.3.1. The continuous permafrost zone

Only 14% of the forested area in the boreal zone is underlain by continuous permafrost, mainly in the unmanaged forest area (Fig. 3). Mineral soils dominate the area and we estimate that they contain 33 Pg C, which is similar in magnitude to the C stock of all the nonfrozen mineral soils (38 Pg) that occupy 52% of the area of the boreal zone (Table 1). This may even be an underestimate given that some of the permafrost soils are of deltaic or paleozoic origin, which contain extraordinarily high organic C stocks matching those found in deep organic soils (Sanborn et al. 2006; Smith et al. 2009; Kanevskiy et al. 2011). The addition of these soil types to a recent re-estimation of C stocks in the northern circumpolar permafrost region contributed to a doubling of previous C stock estimates (Tarnocai et al. 2009). Zimov et al. (2006) reported that the organic matter in similar deposits in Russia decomposed rapidly upon thaw. Therefore, even though these soils occupy a small area of the boreal zone, their contribution to the C budget of the Canadian boreal zone is expected to be significant both nationally and regionally, although little is known about their C dynamics in Canada.

#### 3.3.3.2. The discontinuous permafrost zone (including peatlands)

The discontinuous permafrost zone is where permafrost thaw is of greatest concern to contemporary and future (see section 4.2) C budgets. The majority of the discontinuous permafrost zone lies within the boreal zone. Most of the unmanaged forest area (88%) and nearly half (47%) of the managed forest area is underlain by discontinuous permafrost. Much of the area is dominated by mineral soils with relatively good drainage or as mineral soils within a complex of peatlands with frozen and unfrozen organic soils. Despite their smaller area, the majority of C stocks are found within the organic soils of the peatlands. We estimate that ~41 Pg C are contained in frozen peatland soils and 96 Pg C are contained in unfrozen peatland soils in the boreal zone (Table 1) and the majority (over 90 Pg C) is located in the discontinuous permafrost zone (Bhatti and Tarnocai 2009).

The C dynamics of northern peatlands (frozen or not) have been extensively reviewed over the last decade, particularly in relation to global change (Blodau 2002; Lavoie et al. 2005; Bridgman et al. 2006; Tarnocai 2006). In a recent review, Strack et al. (2008) concluded that northern peatlands are a net long-term sink for atmospheric CO<sub>2</sub> and a source of CH<sub>4</sub>, but some peatlands are sources and others are sinks because of their large spatial variability, and this will likely remain true under changing climatic conditions

(see section 4.2). Carlson et al. (2010) estimated the long-term annual C sequestration rate for all of Canada's peatlands at 23 Mt C but did not consider the impact of fire emissions of C and non-CO<sub>2</sub> GHGs from the burning of peat, which can be significant (Turquety et al. 2007) and, if not included in C budgets, could lead to significant underestimation of emissions, especially in years when large areas burned. Recently, Wania et al. (2009) included permafrost and peatlands in a model applied to the circumpolar region. After introducing permafrost effects, they concluded that NEP was reduced from 1.65 to 0.96 Pg C year<sup>-1</sup> while soil C stocks increased by about 40 Pg. The effect of introducing peatlands as well as permafrost was to double the soil C stock increase to 80 Pg.

Most lines of evidence suggest that the consequence of not including forested permafrost and peatland areas in national-scale accounting is to underestimate C stocks and exclude uncertainty associated with under- or over-estimation of net GHG exchange in response to disturbance or climate change. Readers are referred to section 4 and Price et al. (2013) for a more detailed discussion of processes in the boreal zone that are expected to respond to permafrost thaw and climate change.

### 3.3.4. Forests with bryophyte associations

Bryophytes, predominantly mosses and lichens, are ubiquitous throughout the boreal forest in upland forests and peatlands (Brodo et al. 2001; Turetsky et al. 2010) and their unique role in C and N cycling is described in Turetsky (2003). The physiology and ecology of bryophytes differ from vascular plants (Turetsky 2003; Turetsky et al. 2012) such that feedback between bryophytes and local to regional hydrological and thermal regimes and C dynamics can be significant (Holden 2005; Turetsky 2003). In the context of C dynamics and climate change, the importance of mosses and lichens in peatlands is well studied, but less attention, particularly with respect to C dynamics, has been paid to forests with significant bryophyte associations. These would include peaty, mainly black spruce forests (Lavoie et al. 2005; Bhatti et al. 2006) associated with mosses occurring across the boreal and lichen woodlands that can predominantly be found in the eastern boreal zone (Girard et al. 2008; Venier et al., Manuscript in preparation).

Peaty forests in the Canadian boreal zone occur where the forest floor or organic soil horizons are up to 40 cm thick (storing as much as 190 Mg·ha<sup>-1</sup> C) and almost entirely composed of bryoid material, but are not classified as peatlands, which must have a peat accumulation ≥40 cm thick (Soil Classification Working Group 1998). Although the bryophyte layer is not sufficiently thick to be classified as an organic soil or peatland, it imparts unique ecosystem characteristics interacting with trees and shrubs (Turetsky 2003; Turetsky et al. 2010). Ecosystems of this type commonly occur in the transition zone between upland forest and peatlands and where drainage is poor because of landscape position or because the underlying mineral soil has poor drainage characteristics (Bhatti et al. 2006; Venier et al., Manuscript in preparation). Exclusion of the bryophyte component from the C budget of these forest types will clearly lead to the underestimation of soil C and ecosystem C stocks (Bona et al. 2013) but also to the underestimation of ecosystem-level NPP and R<sub>h</sub>. Turetsky et al. (2010) estimated that mosses in boreal ecosystems can contribute about 20% and 48% of ecosystem productivity in uplands and wetlands, respectively. Across all upland and wetland types, moss NPP ranged from 0.45 to 171 g C m<sup>-2</sup> year<sup>-1</sup> based on a conversion factor of 0.45 for moss biomass to C (Bauer et al. 2006) with average *Sphagnum* subsp. productivity almost three times greater than feather moss productivity (Turetsky et al. 2010). Peaty forests are also important because they are potentially very responsive to disturbance from fire or harvesting and climate change (Hartshorn et al. 2003; Lavoie et al. 2005; Bhatti et al. 2006).

The bryophyte layer can have significant impacts on forest C dynamics. Research over the last decade indicates the need to understand the response of the balance between peat moss (*Sphagnum* subsp.), feather mosses (e.g., *Hylocomium* subsp., *Pleurozium* subsp., and *Ptilium* subsp.), and lichen ground cover to climate change and disturbance impacts to enable modelling of their contribution to the overall forest C budget. Cornwell et al. (2008) concluded that changes in dominance among already coexisting plant groups with different decomposition traits could have a larger impact on the C cycle than the direct impact of climate on decay rates. In the case of bryophytes, it has been shown that the compositional change can affect key ecosystem processes including maximum photosynthetic rate of the forest floor and soil base respiration rates (Bergeron et al. 2009), production, NPP, decomposition rates (Bhatti et al. 2006; Turetsky et al. 2008; Bauer et al. 2009; Fenton et al. 2010), permafrost stability and hydrology (Turetsky et al. 2010), albedo (Bernier et al. 2011), and fuel consumption and fire-patterning behaviour (Shetler et al. 2008). In most cases, the interactions between bryophyte species, hydrology, permafrost, and vascular plants are very complex (Hobbie et al. 2000; Turetsky et al. 2010) and surprising in some instances (Hagemann et al. 2010; Bernier et al. 2011).

Inclusion of the bryophyte component of peaty forests in national-scale forest C budgets and forest ecosystem models (Bond-Lamberty et al. 2007a) is a challenge because we do not know their distribution and extent at a national scale (Bhatti et al. 2002; Lavoie et al. 2005), mapping is difficult (Rapalee et al. 2001), and we have insufficient but growing knowledge of their ecology and response to disturbance (Lavoie et al. 2005). Including bryophyte contributions to peaty forests and lichen woodlands in models requires improved data on their spatial distribution in relation to forested and nonforested areas, abundance, type, productivity, interactions with decomposers, and decomposition rates, as well as their indirect impacts on soil thermal regimes and interactions with hydrological regimes.

### 3.3.5. Harvesting impacts on soil carbon

Clear-cut harvesting produces a short-term pulse of slash and other DOM (stumps and roots) and reduces the annual input of biomass C (foliage, fine roots, and other biomass turnover) to DOM pools (see section 2.1). The net C balance of DOM and soil pools with the atmosphere is often negative for some years after harvest, leading to net reductions in litter, dead wood, and soil C pools. The effect of harvesting on C transfers between pools and their subsequent decomposition is reflected in the NFCMARS, but its impact on C stock changes remains the subject of ongoing research.

The frequently cited decline in forest soil C stocks in response to harvesting was primarily attributed to the observations of Covington (1981) in a temperate forest ecosystem, but Yanai et al. (2003) were not able to verify the predictions upon remeasurement of the same sites. However, Nave et al. (2010) concluded from a recent meta-analysis that, on average, harvesting caused temperate forest floor C to decline by a consistent 30% ± 6%. Variability in responses was high and explained mainly by soil taxonomic order, species composition, and time since harvest. Similar meta-analyses have not been conducted for the Canadian boreal forests, which differ significantly from temperate forests in species composition, soil taxa, NPP, decomposition rates, and harvesting practices.

Soil C stocks in boreal systems are typically higher than those in temperate systems because site conditions in the boreal forest are often conducive to accumulation of soil C (Wieder et al. 2006; Jandl et al. 2007; Bhatti and Tarnocai 2009) and R<sub>h</sub> is low for boreal relative to temperate ecosystems (Pregitzer and Euskirchen 2004). Bhatti and Tarnocai (2009) estimated that, in boreal ecozones, C stocks range from 37.4 to 55.4 Mg·ha<sup>-1</sup> for



organic soil horizons and from 93 to 213 Mg·ha<sup>-1</sup> for the underlying mineral soil. These values are higher than those estimated by Nave et al. (2010) for temperate forests (5–50 Mg·ha<sup>-1</sup> organic horizons; 5–80 Mg·ha<sup>-1</sup> for mineral soil). A recent review (Thiffault et al. 2011) comparing whole-tree and stem-only harvest impact in boreal and temperate forests in Canada concluded that C-poor soils (small pool size) were most sensitive to whole-tree harvest. However, when pool size is small (temperate and some boreal forest floor C stocks), the loss of a small amount of C can translate into a high proportional loss (Nave et al. 2010). Applying such high proportional losses to the boreal zone where forest floor C stocks are mostly large could lead to the potentially erroneous conclusion that harvesting in the boreal results in large losses of soil C to the atmosphere.

The high proportional loss of soil C in response to harvesting estimated for temperate systems may not occur in some of the dominant forest types of the Canadian boreal zone. This is especially true for black spruce forests, the most common coniferous forest cover type in the boreal forest (Lavoie et al. 2005). The dominant natural disturbance type in black spruce systems is wildfire, and consensus is emerging that harvesting has a less negative impact on the C budget of black spruce forests compared with wildfire (Bergeron et al. 2008). This is primarily because harvesting practices in most black spruce forests are less disruptive than wildfire (Amiro et al. 2001a), as harvesting is often restricted to the winter when the ground is frozen and covered in snow, thus, avoiding large disturbance to the soil (Lavoie et al. 2005). Studies examining black spruce stands in Ontario and Quebec generally indicated no response of mineral soil C to harvest disturbance. In some cases, reduction in forest floor C in younger stands was attributed to a change in harvest practices during the past several decades from horses to more disruptive mechanical logging (Brumelis and Carleton 1988) or post-harvest burns (Scheuner et al. 2004) rather than increase in decomposition rates. In black spruce systems conducive to paludification (a shift from non-peatland to peatland caused mainly by a change in the hydrologic balance to wetter conditions) (Fenton et al. 2010), low-impact harvesting can create conditions that favour C accumulation rather than loss (Lavoie et al. 2005).

Two emerging themes in temperate forest soil C research that may affect this conclusion are (i) most studies to date focus on C stock changes in the surface soil and do not account for the response of soil C at depth (to 100 cm or greater) and (ii) the apparent stability of mineral soil C may change in response to change in the environment, which can occur because of harvesting (Harrison et al. 2011). However, no research, to our knowledge, has been conducted in the Canadian boreal forest to study C dynamics at depth in response to harvesting or distinguish C that has accumulated from that which is stabilized (Jandl et al. 2007). In particular, we know little of the degree of stabilization of C (Marschner et al. 2008) in Canadian boreal forest soils (Norris et al. 2011).

### 3.3.6. Invasive earthworms

Since the time of Darwin (1881) it has been known that earthworms are important agents of soil formation and nutrient dynamics. However, the Canadian boreal forest zone has evolved in the absence of this significant ecosystem engineer (Evers et al. 2012). Non-native, primarily European, earthworms are being introduced to areas of boreal forest following the regional extinction of native species during the last ice age (Addison 2009). The primary vectors of spread are associated with human recreational and resource development activities (Cameron et al. 2007), the rate of which are expected to increase in the coming decades. Earthworms can have large impacts on GHG emissions (Lubbers et al. 2013) and DOM and soil C dynamics (Langor et al., Manuscript in preparation). They can reduce forest floor C stocks either through an increase in decomposition rates or transfer rates to

the mineral soil (Langmaid 1964; Hale et al. 2005) and can stabilize C in the mineral soil (Shaw and Pawluk 1986). Reduction in forest floor C stocks by earthworms has implications for estimation of C emissions from fire that originate mainly from the combustion of the forest floor (Letang and de Groot 2012).

Although a model of the effects of earthworms on soil C was recently developed for temperate forests (Huang et al. 2010), it has not been adapted or tested for the boreal zone. Further to this, effects of earthworms are currently not included in landscape-scale models of boreal forest C dynamics because of insufficient data on the spatial distribution and rates of spread of earthworms in the boreal zone of Canada and because of an inadequate understanding of their effects on net C fluxes in boreal forests. However, given their site-level effects on forest floor and mineral soil C dynamics and their expansion in many parts of the boreal zone, the omissions of earthworm impacts in national-scale analyses of forest C budgets could contribute significant uncertainties to present and future estimates.

## 4. Future projections of Canada's boreal forest C cycle

The future C balance of the Canadian boreal forest will affect the global atmospheric C budget and influence the level of global mitigation efforts required to attain atmospheric CO<sub>2</sub> stabilization targets (Allen et al. 2009). This goal will be easier if forests continue to globally act as C sinks (Pan et al. 2011a). However, the potential for large increases in emissions from boreal forests and other terrestrial systems is real (e.g., Lavoie et al. 2005; Boisvenue and Running 2010; Metsaranta et al. 2011; Schuur and Abbott 2011) and would contribute to increases in atmospheric CO<sub>2</sub>. Human-induced changes to the global environment have already affected forest systems (Boisvenue and Running 2006; Kurz et al. 2008c; Allen et al. 2010; Hember et al. 2012), and environmental changes are projected to intensify (IPCC 2007). The complexity and diversity of ecosystems combined with the range of environmental changes will result in regions and time periods with positive and negative feedbacks (i.e., net sources or net sinks) (Le Quéré et al. 2009; Boisvenue and Running 2010) and potentially large changes in the net balance over time (Morales et al. 2007; Xiao et al. 2010; Metsaranta et al. 2010). This section addresses future changes in the key drivers affecting the C balance in Canada's boreal forest, which include forest responses to environmental changes, changes in disturbance regimes, and changes in human activities and land use. For further review of future changes in the boreal zone, see Price et al. (2013).

### 4.1. Changes in forest dynamics

Productivity increases are already reported owing to warmer temperatures and longer growing seasons, increased atmospheric CO<sub>2</sub> concentrations, and N in temperate (Hember et al. 2012) and boreal forests (Magnani et al. 2007; Briffa et al. 2008; Hickler et al. 2008). A recent review of measured changes in forest productivity globally found that 75% of papers reviewed reported increased productivity and the remaining studies declining (10%), mixed (8%), or no trend (4%) in productivity (Boisvenue and Running 2006). Other reports of changes already occurring were outlined in section 3.3.1, and several process modelling and experimental studies, some conducted in Canada's boreal forest, project future changes to growth rates (Chen et al. 2000; Gamache and Payette 2004; Hickler et al. 2008; Silva et al. 2010; Zhao and Running 2010; Beck et al. 2011; Berner et al. 2011; Toledo et al. 2011).

Factors that could potentially limit the growth response include moisture constraints, nutrient availability, thin soil cover, and (or) pest disturbances (Jarvis and Linder 2000; Lafleur et al. 2010; Beck et al. 2011). Responses to environmental change will vary by species (Cole et al. 2010), ecological region (Paquette and Messier 2011), provenance (McLane et al. 2011a), and management regime

(Cyr et al. 2009). Drought (Allen et al. 2009; van Mantgem et al. 2009; Zhao and Running 2010), disease (Sturrock et al. 2011), insects (Hicke et al. 2012) or genetic adaptation (McLane et al. 2011b) may decrease future productivity or increase future mortality (Peng et al. 2011; Ma et al. 2012) or both. Projected changes in disturbances patterns that may affect the C balance of boreal forest are discussed in a subsequent section.

Many C budget projections for Canada's boreal forest assume that the distribution of tree species does not change over time (e.g., B. Smith et al. 2011). Climatically suitable ranges for tree species will likely exhibit large-scale redistribution over the 21st century (e.g., Hamann and Wang 2006; McKenney et al. 2007; Schneider et al. 2009). Species ability to shift their distribution to spatially track changes in climate will depend on dispersal ability (Greene et al. 1999; Nathan et al. 2011), competition (Loehle 1996), and soil conditions (Lafleur et al. 2010), as well as interactions among these factors (Leithead et al. 2010) and with disturbances (Greene and Johnson 2000; Greene et al. 2004; Simard and Payette 2005; Johnstone and Chapin 2006a, 2006b) and regeneration success (Classen et al. 2010). The future distributions of tree species are highly uncertain and depend strongly on the assumptions of the model used to make the projection (Loehle 2000; McKenney et al. 2007; Morin and Thuiller 2009; Mbogga et al. 2010). It is unlikely that species will be able to migrate as quickly as their climatically suitable ranges change (Aitken et al. 2008), resulting in local populations that are progressively more genetically maladapted to their new climate. If unable to migrate, species will only persist locally through genetic adaptation to new conditions or face extirpation (Aitken et al. 2008; Barnes 2009). Inter species ability to adapt also differs (Trindade et al. 2011). Over the 21st century, the response of trees to shifts in climatic niches will likely be a combination of gradual change in areas where seed dispersal limits distributions and rapid shifts to new ecosystem states where thresholds are surpassed (Chapin et al. 2004). Along with maladaptation and mortality or a gradual or sporadic change in species will come a change in productivity and, hence, a change in C balance.

Carbon stocks may also be influenced indirectly by changes in the distribution of boreal forest relative to other ecosystem types (e.g., Beck et al. 2011). The southern boundary of the western Canadian boreal forest is a forest–grassland ecotone. As a result, there is a higher risk that these transitions will result in a shift to nonforest communities, either grassland typical of the prairies or sparse woody vegetation currently typical of the aspen parkland (Hogg and Hurdle 1995; DeSantis et al. 2011). In northern Quebec and other parts of the Canadian boreal zone, repeated fires can lead to a transition from closed-crown forests to lichen woodlands with possible reduction (Girard et al. 2008) or possible increases (Lavoie et al. 2005) in C stocks but also altered albedo (Bernier et al. 2011). The southern boundary of the eastern Canadian boreal forest is typically a boreal–temperate forest ecotone (i.e., hemiboreal subzone, Brandt 2009), where transitions are likely to result in a shift in dominance from boreal to temperate tree species representative of Great Lakes St. Lawrence or tolerant hardwood forests, rather than a loss of forest cover (Leithead et al. 2010). The influence of these effects on C dynamics is still uncertain. A transition to grassland would result in a reduction in C stocks; but a change in tree species could affect C stocks in either direction, depending on the new species and soil C dynamics.

Climate warming can also lead to changes in the dynamics of vegetation at treeline (Körner 2012). Increased tree recruitment at the forest–tundra ecotone can result in an advance of both altitudinal and latitudinal treeline into areas of tundra (e.g., Gamache and Payette 2005). A recent global review shows that about half of the studies examining changes over the last century have recorded an advancing treeline (Harsch et al. 2009). Northern forest extension could lag warming for a few decades and transient species associations may initially develop, but over time forest limits

could advance to those experienced before the Little Ice Age (MacDonald et al. 2008). Climate warming can also lead to increases in vegetation density, particularly shrub vegetation, in the northern boreal forest and the Arctic, which has been demonstrated by remote-sensing studies (e.g., Sturm et al. 2001; Pouliot et al. 2009; Fraser et al. 2011; McManus et al. 2012), also referred to as vegetation greening. Figure 10 shows an example of this process (NASA 2012a). A thorough review of the processes of shrub vegetation thickening and advance, including both promoting and limiting factors, is provided by Myers-Smith et al. (2011). The total impact of treeline advance and vegetation greening in the Arctic on future boreal forest C dynamics is difficult to predict, but their main impact will be on the relative distribution and extent of forest and tundra vegetation. There are also complex interactions between factors such as local environmental conditions (e.g., Mamet and Kershaw 2013), fire disturbance (e.g., Arseneault and Payette 1992), and their interactions (e.g., Munier et al. 2010) that still remain to be completely understood.

Process modelling of environmental impacts on growth rates needs to be linked to modelling of ecosystem and species shifts across ecologically complex regions to help assess the effects of opposing factors in forest dynamics and the C balance of these forests. Modelling capabilities are improving but are not yet reaching the ability to estimate future forest productivity. There is presently no agreement on the direction, magnitude, or cause of net changes in the future productivity of Canada's boreal forest (e.g., Bernier et al. 2010; Shanin et al. 2010).

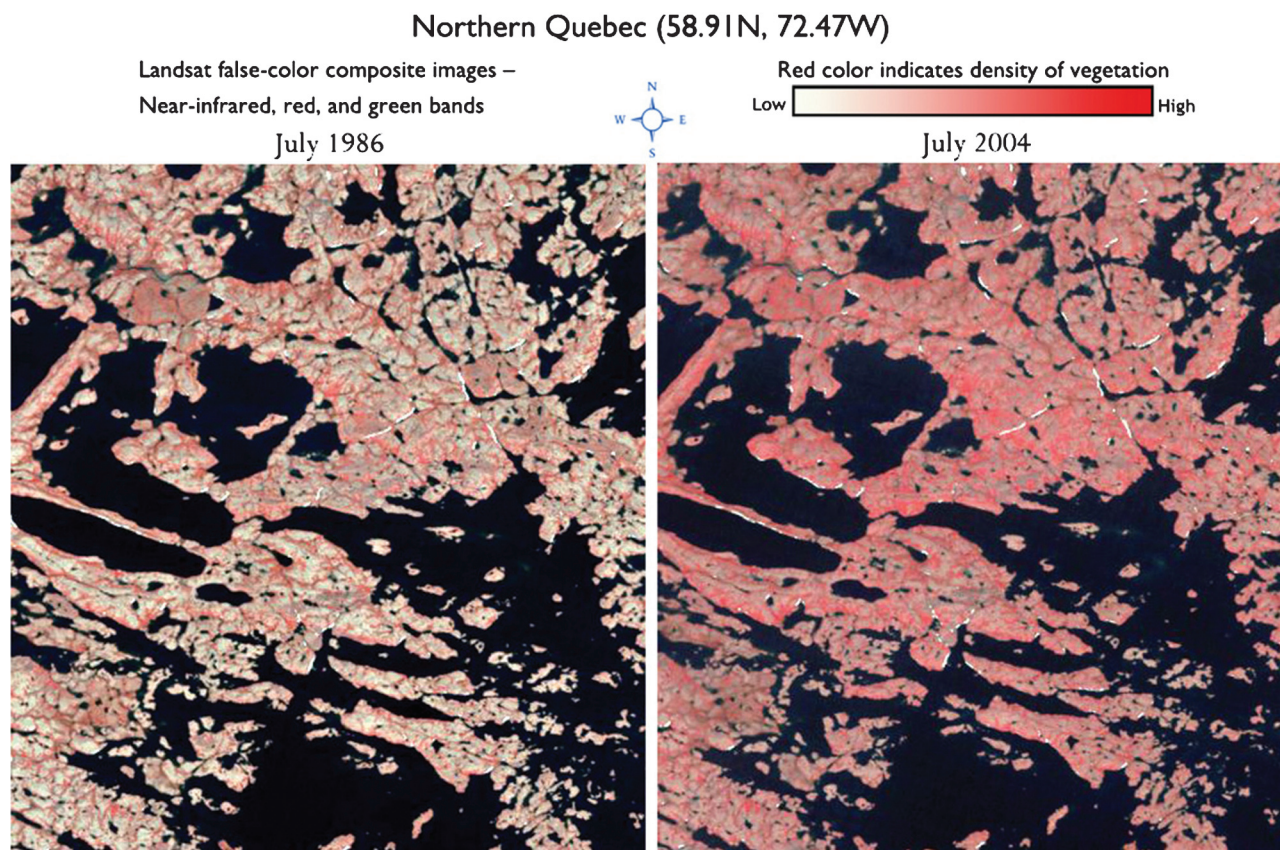
#### 4.2. Changes in decomposition rates

Greenhouse gas emissions to the atmosphere from boreal soil C are expected to increase with future warming because of the temperature sensitivity of soil organic matter decomposition that is commonly used in models (Rodrigo et al. 1997; Peltoniemi et al. 2007) and the presumption that soil temperature will increase apace with air temperature. Predicted higher air temperatures are also expected to affect NPP and so it is the difference in the rates of changes in NPP and  $R_h$  that will determine the change in NEP.

The degree to which soil decomposition is sensitive to temperature remains uncertain, with some studies suggesting higher sensitivity (Knorr et al. 2005) and others lower (Davidson et al. 2000; Giardina and Ryan 2000) sensitivity to temperature increase. Experiments have found a range in temperature sensitivity with  $Q_{10}$  values from 1 to 4 (Boone et al. 1998; Irvine et al. 2005; Paré et al. 2006; Gaumont-Guay et al. 2008; Fissore et al. 2009). Conflicting or unexpected responses of the temperature sensitivity of  $R_h$  may be observed because studies measuring and modelling decomposition do not directly represent the soil (e.g., microbial activity, stabilization with inorganic components, and aeration) and landscape (e.g., permafrost thaw, thermokarst, change in insulation from peat, forest or snow cover, and hydrology) processes controlling  $R_h$ . However, these processes may have counteracting effects (Davidson and Janssens 2006; Conant et al. 2011) and respond differently to temperature change. Recognition that modelling of soil C dynamics may be more complex than previously thought (Dungait et al. 2012) has led soil C modelling researchers to propose changes to the fundamental approach to modelling soil C and its response to climate change (Wutzler and Reichstein 2008; Allison et al. 2010; Conant et al. 2011; Schmidt et al. 2011). Suggestions include modelling of soil C should abandon the structure of pools with intrinsic decomposition rates, including the vaguely defined “recalcitrant” fraction, and should instead move towards the representation of processes directly controlling  $R_h$  (Conant et al. 2011; Schmidt et al. 2011).

Aside from uncertainties associated with the sensitivity of decomposition to temperature, we also need to understand the magnitude and trajectory for boreal soil temperature (or soil climate) change in response to climate change. Long-term change in soil

**Fig. 10.** A comparison of area in northern Quebec showing increased vegetation density between 1986 and 2004 (credit: Jeff Masek). Available from <http://www.nasa.gov/topics/earth/features/shrub-spread.html> [accessed 10 September 2012].



temperature is poorly documented but studies in Canadian (Zhang et al. 2005) and European boreal zones (Helama et al. 2011) have shown that it will not necessarily exhibit the same trends in air temperature as is often presumed in C models. The discrepancy between air and soil temperature and their shifts with climate change in the boreal is likely attributable to spatially and temporally complex interactions between edaphic and landscape factors affecting hydrology, permafrost, snow cover, peat, and vegetative cover (Bond-Lamberty et al. 2007a; O'Connor et al. 2010; Hennon et al. 2012). The response of these factors to climate change affects the trajectory and rate of change in soil climate (including temperature, moisture, and aeration — oxic versus anoxic condition) for decomposition.

Permafrost thaw is a process of great concern in the discontinuous permafrost zone of the boreal zone (Fig. 3) because it has the potential to significantly contribute to atmospheric feedbacks and climate change (Schuur and Abbott 2011). It not only changes soil climate for decomposition but also the size of the C pool available for decomposition because the C is no longer frozen or permafrost thaw has altered the water table and the proportions of C under oxic and anoxic conditions. The movement of C from frozen to unfrozen state may proceed at rates an order of magnitude higher than the direct effect of temperature sensitivity (Schuur et al. 2009). Once thawed, the rate of permafrost C decomposition may be very high in the cases where C stability was low at time of freezing (Zimov et al. 2006) or relatively low for some peatland types if the organic matter was already relatively decomposed at the time of freezing (O'Connor et al. 2010). Recent studies have suggested that, during the initial phases of permafrost thaw, uptake of C resulting from increases in plant productivity can initially compensate for loss of C due to increases in  $R_h$ , but that a tipping point will be reached where  $R_h$  exceeds NPP and that the

resulting C emissions are expected to continue into the future (O'Donnell et al. 2012; Schaphoff et al. 2013). Tarnocai (2006) rated the sensitivity of peatlands to climate change into six classes ranging from “no change” to “extremely severe”. We estimate that frozen peatlands in the severe and extremely severe sensitivity classes contain 34 Pg C in the boreal zone (Table 1). Tarnocai (2006) also estimated that 36 Pg C in unfrozen peatland is at equal risk and some portion of these peatlands also occurs in the discontinuous permafrost zone. However, it is also this zone that presents the most challenge to models used to predict the response of permafrost thaw to climate change (Schaefer et al. 2011) because of the models' limited ability to represent small-scale heterogeneity and feedbacks that can lead to both positive and negative effects on permafrost stability (Strack et al. 2008; Grosse et al. 2011; Schaefer et al. 2011). Schaefer et al. (2011) reported on model estimates for permafrost degradation in the 21st century that range from 16% to 85% with their own model predicting a 29%–59% reduction. Caution must be exercised, however, when applying these rates to the Canadian boreal zone because they are largely derived from permafrost areas outside of the Canadian boreal zone (global arctic and subarctic zones) and have the common challenge of representing the heterogeneity of the discontinuous permafrost zone that dominates the boreal zone. Although it is generally accepted that permafrost and peatland soils in the Canadian boreal zone will be sensitive to climate change (Tarnocai 2006; Zhang et al. 2008), it is still unclear if the outcome will be a net increase in productivity or emissions to the atmosphere as  $\text{CO}_2$  and (or) methane ( $\text{CH}_4$ ) (Lavoie et al. 2005; Turetsky et al. 2007; Yu et al. 2011). Uncertainties are particularly high for the prediction of  $\text{CH}_4$  emissions from peatland and permafrost thaw in response to climate change (O'Connor et al. 2010), and some of the processes important to the cycling of methane are just being dis-

covered. For example, recent research (Kip et al. 2010) found that CH<sub>4</sub> released from the decay of sphagnum mosses can be oxidized by symbiotic methanotrophs and the C reassimilated by the moss when submerged, which provides a mechanism to potentially reduce CH<sub>4</sub> emissions.

In forested nonpermafrost zones, increases in NPP with climate change can result in higher inputs of C to soil from foliage and fine root turnover (Matamala et al. 2003; Iversen et al. 2008), but this does not necessarily result in an increase in soil C (Schlesinger and Lichter 2001). Tree species have a strong influence on root allocation responses to CO<sub>2</sub> increase (Matamala et al. 2003), and whether or not R<sub>h</sub> is stimulated by increased inputs depends on soil characteristics (Bader and Körner 2010). Priming of soil respiration by greater inputs of root and foliar C may result in greater respiration of older soil C (Trueman and Gonzalez-Meler 2005; Fontaine et al. 2007). Understanding and modelling landscape-level hydrology is critical to predicting the C budget of the most northerly unmanaged forest area (see section 3.2.2) and also important in the more southerly managed forest area that may be unaffected by permafrost but where topographic controls on drainage patterns influence soil C stocks (Webster et al. 2011) and soil respiration (Webster et al. 2008).

#### 4.3. Future disturbances

Carbon dynamics in the Canadian boreal forest have historically been dominated by natural disturbances, with fire as the dominant disturbance in the western boreal forest (Fig. 4; Bergeron et al. 2004; Lavoie et al. 2005; Balshi et al. 2007) and both fire and insects affecting the eastern boreal forest (Blais 1983). Recent changes in climatic conditions have contributed to increased impacts of drought and insects (Peng et al. 2011). Positive feedback to climate change could result from increasing disturbance frequency and intensities. This section reviews some of the potential future effects of these disturbances on the C balance of the boreal forest. For additional information, see also Price et al. (2013).

Projected future weather conditions in the Canadian boreal forests increase the probability of fire occurrence (and hence of area burned) over the 21st century (Flannigan et al. 2005b, 2009; Balshi et al. 2009; Krawchuck et al. 2009; Hély et al. 2010). Increases in fire and other disturbances will contribute to increased emissions and forests will (other things being equal) store less C, thus contributing towards “positive feedback” to climate change (Metsaranta et al. 2010; Melillo et al. 2011). However, these effects are not likely to occur uniformly. Area burned in Canada’s boreal forest fluctuates widely from year to year (Armstrong 1999; Amiro et al. 2001b; Stocks et al. 2003). As a result, detecting changes in fire regimes from short time series of data are almost impossible (Metsaranta 2010). In addition, much of the cumulative area burned over a given period of record occurs in a small number of years with large area burned, and the frequency and magnitude of these extreme fire years is also highly uncertain (Metsaranta 2010). Vegetation succession that increases the proportion of deciduous forests in the boreal forest region could provide a negative feedback with respect to the projected increase in fire (Johnstone et al. 2011). However, quantitative studies on the selectivity of burning behaviour with respect to forest type in Canadian boreal forests are inconsistent. Cumming (2001) supports the hypothesis that coniferous forests burn more than their proportional contribution to landscape composition, but Podur and Martell (2009) suggest that all forest types burn in proportion to their composition.

Furthermore, timing of the fire has been shown to have an effect on the depth of burn, with late-season burns resulting in more of the ground surface organic matter consumed (Turetsky et al. 2011), which has significant consequences on the C balance. Such conditions could also allow extreme fire events with pro-

longed smoldering phases (and the associated high CH<sub>4</sub> emissions) under snow and during winter months.

Impacts are expected to be greater in the drier, continental western boreal than the eastern Canadian boreal forest (Flannigan et al. 2005b; Balshi et al. 2009), and increased fire occurrence in the future is projected to overwhelm the capacity of fire-management agencies to mitigate these effects through increased suppression effort (Flannigan et al. 2005a; Podur and Wotton 2010). Projections of the impact of increases in area burned in all of Canada’s managed forest over the 21st century, assuming that area burned would increase by a factor of 4 in most of western Canada and a factor of 2 in eastern Canada, suggest that to maintain ecosystem C stocks NEP would have to increase by about 25% to offset increased fire emissions. An increase in NEP of this magnitude, sustained over time and over all areas in which disturbances increase, is not likely (Kurz et al. 2008a; Metsaranta et al. 2010).

Climate change is predicted to affect future insect disturbances in several ways: range expansions northward and to higher elevations (Régnière et al. 2010; Safranyik et al. 2010); increased temperature allowing insects to mature more quickly, reduce winter mortality, and increase summer productivity (Raffa et al. 2008); and changes in the synchrony of insect life cycle stages and plant phenology (Nealis and Régnière 2004; Régnière et al. 2009). The net impacts of these changes on the forest C balance are difficult to predict, but insect outbreaks can have large impacts on C stocks and fluxes (Kurz et al. 2008c; Dymond et al. 2010; Hicke et al. 2012). The main impacts of insects in the boreal forests of Canada have historically been confined to the southern regions, and the potential for range expansion into regions where host tree species are present but have historically not been challenged by insects could result in increased tree mortality and greater reduction in C stocks. Examples include forest tent caterpillar in the Northwest Territories in 1995 and 1996 (Brandt 1997) and the potential spread of mountain pine beetle on jack pine across Canada’s boreal zone (Safranyik et al. 2010).

Parts of the boreal forest, particularly in western Canada (Michaelian et al. 2011), but also in parts of eastern Canada (Hély et al. 2010), are expected to experience more frequent and severe droughts in the 21st century, potentially impacting several ecosystem processes that influence forest C dynamics (van der Molen et al. 2011). Increases in drought-induced forest mortality have already been observed globally (Allen et al. 2010; van Mantgem et al. 2009; Huang and Anderegg 2012) and in Canada (Michaelian et al. 2011; Peng et al. 2011; Ma et al. 2012). Precipitation is found to influence forest productivity in both tree-ring (Beck et al. 2011) and flux tower (Schwalm et al. 2010) studies. In addition, drought can influence soil C dynamics, with dry conditions potentially resulting in reduced decomposition rates (Allison and Treseder 2008; Smyth et al. 2010) that under some conditions can offset C balance impacts resulting from productivity losses. The net impact of these effects on the C balance of boreal forests in Canada has not yet been quantified. Most likely, impacts will vary by forest types and regions depending on moisture regimes and site conditions, as well as interactions with insects and pathogens. Deciphering the physiological mechanism by which trees decline and die under drought will soon contribute to better modelling and prediction of drought events and their effects on C balance (Anderegg et al. 2012).

#### 4.4. Land-use change

Future economic, social, and climatic conditions will affect the future rates of deforestation across the boreal zone. In the northern boreal forest, large individual events and developments are anticipated to have the main impact. For example, several hydroelectric developments are being considered over the next 25 years in northern Manitoba, Quebec, Labrador, and British

Columbia. Construction of new mines and their access roads, programs to connect northern communities to the electric grid and permanent roads, and other efforts to improve access to northern resources are all expected to contribute areas of deforestation.

The main forestry access road system in the boreal forest is becoming largely developed but is expected to continue to move northward into previously unaccessed areas of commercially harvestable forest. The oil and gas infrastructure is also expected to expand where resources have been found. In the oil sands regions, increased use of subsurface steam-assisted gravity drainage (SAGD) methods for oil extraction will contribute deforestation from well pads, pipeline, access roads, and steam generating and processing facilities. Activity in the Northwest Territories is also increasing, and the construction of the Mackenzie Valley pipeline will result in areas of deforestation. Within the mineable oil sands district, considering the total mineable area, its forest cover, planned developments, and expert opinion on future development, a reasonable estimate of the total area eventually disturbed is in the order of 200 kha of which approximately 75%–85% will involve forest.

The decline in deforestation for agricultural land conversion is expected to continue in the future, although changes in economies, demand for agriculture-based biofuels, and government policy under climate change could also result in agricultural expansion into forest regions. Most land currently under agriculture in the boreal zone is capable of being afforested. Future afforestation rates are expected to be low and similar to current rates, in the absence of government incentive programs (Lemprière et al. 2013). In the boreal Clay Belt regions of Quebec and Ontario, considerable abandonment of agriculture land has occurred over the past decades. Some of this will eventually revert to forest; and where these forest areas are included in provincial inventories, the C consequences are captured in the regional C balance estimates. It is unknown whether abandonment of agriculture land in the Clay Belt will continue in the future or whether this land will be reclaimed for agricultural uses, but growing world population, higher demand for food, and raising food prices all increase the pressure to reclaim land for agricultural uses.

#### 4.5. Net carbon balance

The future net balance of C emissions and removals in Canada's boreal forest will be affected primarily by changes in forest productivity (section 4.1), decomposition rates (section 4.2), and natural disturbances (section 4.3). While it remains impossible to predict with certainty the resulting net C balance under future climate conditions, the "asymmetry of risk" (Kurz et al. 1995) is of concern: for boreal forests stands to reach maturity and maximum C storage, many decades of survivable growing conditions must prevail, but it takes only a single extreme event such as drought, windthrow, fire, insects, or other disturbances to kill trees or stands. With climate change predictions including more frequent extreme climatic events (Rahmstorf and Coumou 2011; Hansen et al. 2012), increases in natural disturbances (Flannigan et al. 2005b; Balshi et al. 2009), and maladaptation of forest ecosystems to shifting climate conditions (Aitken et al. 2008; Barnes 2009), the probability that boreal forests C stocks will increase under climate change scenarios is lower than the probability that they will decrease.

A second argument for the likely decrease in boreal forest C stocks with climate change is that considerably more C is stored in DOM and soil C pools than in biomass, largely because cold, wet, anoxic, and frozen environments have delayed or prevented decomposition. As discussed previously, warming is generally predicted to increase decomposition, leading to reductions in DOM and soil C pools that are likely greater than a possible increase in biomass C pools, resulting in an anticipated net decrease in boreal forest C stocks.

#### 5. Knowledge gaps and monitoring needs

The net C balance of Canada's boreal forest is dominated by two large fluxes: NPP and  $R_h$ , processes that continuously occur in all forest ecosystems. Any changes in environmental conditions such as climate change,  $CO_2$  fertilization, and N deposition that affect NPP and  $R_h$  have the potential to cause large changes in the net C balance, in particular if environmental changes have opposing impacts on the two fluxes, e.g., decreasing productivity and increasing respiration losses. In addition to the continuous processes of growth and decomposition, some ecosystems are in some years affected by anthropogenic and natural disturbances, and the boreal forest C balance will be strongly affected by changes in disturbance regimes, i.e., the frequency, intensity, and types of disturbances.

To reduce the uncertainties in the estimates of the current and future C balances of the boreal forest, research needs to be directed to improve understanding of (i) continuous processes determining NPP and  $R_h$ , (ii) disturbance-related processes, (iii) interactions between disturbance and ecosystem production, and (iv) interactions among landscape distribution of forests, environmental drivers of disturbance, and successional trajectories.

Models of forest C dynamics have been used successfully to derive estimates of net C balances that take into account the broad distribution of forest characteristics including species, forest age, site conditions, and the impacts of natural disturbances and forest management. By necessity, such models incorporate assumptions about homogenous conditions within forest stands and landscapes. Although the spatial resolution of models used at the scale of the boreal forest has increased by more than three orders of magnitude over the past 20 years, the "average" stand represented by such national-scale models today is typically about 100–1000 ha. Depending on vegetation characteristics, topography, and soil conditions within average stands of such size, the ecological processes that determine C fluxes can be occurring at a range of rates and respond differentially to environmental changes. Further improvements in (i) the spatial resolution of modelling approaches down to 1 ha resolution and (ii) the availability of spatially-explicit data on forest characteristics, topography, and soils at the increased resolution have the potential to contribute to reducing uncertainties of C stock and C flux estimates, provided that sufficient data on environmental characteristics are available at that fine spatial scale (Canadian Forest Service, Natural Resources Canada 2012). Many forest ecosystems models operate on annual time scales, and increasing the temporal resolution would allow the improved representation of processes occurring at seasonal, monthly or daily time scales. Efforts to reduce uncertainties in C budgets by increasing spatial and temporal resolution of models will substantially increase the demands for input data and computing resources.

Improvements in remote-sensing techniques combined with forest ecosystem models will likely achieve further reductions in the uncertainty of disturbance-related C flux changes in the coming years. In contrast, reducing uncertainties about subtle changes in fluxes in response to fluctuating environmental conditions will remain an ongoing challenge. Every 1 g m<sup>-2</sup> year<sup>-1</sup> change in net fluxes over Canada's boreal forest sums to a change of 2.7 Tg year<sup>-1</sup> (or nearly 10 × 10<sup>6</sup> Mg year<sup>-1</sup> of CO<sub>2</sub>) in the boreal forest C balance; thus, even subtle changes in fluxes, undetectable with currently available methods, occurring in synchronicity over large areas can have large impacts on the global C cycle. Eddy covariance flux towers have successfully been used to quantify high-frequency flux responses to environmental drivers, but measurements have been limited to a small number of sites, each with a small footprint and relatively short observation period. Thus, spatial and temporal upscaling of ecological processes remains a major challenge.

New techniques to quantify changes in growth and mortality rates in response to environmental change using tree-ring data

(Metsaranta and Kurz 2012) and permanent sample plots (Hember et al. 2012) offer opportunities to gather empirical data on forest responses to climate change over much larger areas and multi-decadal time periods. Such data can help inform and constrain process models. Combined with site-specific ecosystem process models, e.g., ecosys (Grant et al. 2007), InTec (Chen et al. 2000), or 3PG (Landsberg and Waring 1997), these data offer a path to further reducing uncertainties in carbon flux estimates (Keenan et al. 2012). However, as climate change continues to affect Canada's boreal forests, ongoing monitoring of forest growth and mortality responses to climate change will be required. In recent years, the numbers of climate-monitoring stations, permanent sample plots, and flux towers in Canada's boreal forest have all decreased while the need for monitoring data has increased, and the ability to extract scientifically relevant knowledge from such measurements has improved.

Traditional forest inventories, permanent sample plots, tree-ring analyses, and meta-analyses provide significant insights into forest responses to environmental changes. But, in boreal forests, a larger proportion C is stored in DOM and soil C pools for which much fewer measurements exist. Moreover, considerable uncertainty remains on the impacts of climate change on  $R_h$  (Pendall et al. 2004; section 4.2). Research needs to reduce this knowledge gap include linking different agents of tree mortality to fall and decay rates, time series of soil C stock measurements using consistent methodologies to enable the detection of trends in C stocks in response to environmental changes, monitoring of changes in permafrost distribution and active layer depth, soil-warming experiments to better understand processes that will change in the future, transect studies along climate gradients, and the quantification of soil C dynamics.

Effects of increased atmospheric  $CO_2$  on aboveground production have been investigated, but effects on belowground processes have received much less attention and as a result are not well understood (Iversen et al. 2008). Carbon dioxide fertilization responses could cause changes in deep soil C pools, for example, through changes in C allocation to fine roots, through increased production, and through allocation of fine roots to deeper layers in the soil profile. Therefore, plot data on fine root production, its vertical distribution in the soil profile, and data on turnover rates are needed to help quantify the effects of increased atmospheric  $CO_2$  concentrations and climate change (e.g., Olesinski et al. 2012).

Gaps also remain in the representation and quantification of processes and pools. The contributions to Canada's boreal C cycle of bryophytes, deep organic soils, and permafrost thawing are three examples of areas in which insufficient understanding and quantification at the national scale contribute significant uncertainties to the estimates of pools, their current C balance, and their projected future changes.

Reduction in uncertainties of regional- and national-scale estimates will require models that integrate environmental and climate data, forest inventories, and information obtained from remote sensing to scale up site-specific knowledge to larger areas and over longer time periods. Data assimilation approaches that combine ecosystem models and empirical data and constrain flux estimates using plot-level data, EC tower data, remote-sensing information, and inverse-modelling approaches offer new methods to reducing uncertainties (Richardson et al. 2010a; Chen et al. 2011). Remote-sensing techniques are increasingly detecting large-scale changes in forest reflectance properties that are correlated with forest productivity (Zhao and Running 2010; Beck and Goetz 2011). While the interpretation of such observations remains under development, opportunities exist to improve the scientific understanding of remotely sensed responses through comparisons against ground observations, including permanent sample plot data, tree-ring measurements, and flux tower measurements, as discussed previously.

Geological and edaphic conditions may be extremely variable even at very fine spatial scales, yet these have tremendous impact on C cycling (Ju et al. 2006). In particular, soil water and drainage needs to be better known to estimate the C balance of high latitude ecosystems (Ju et al. 2010). Predicting hydrological patterns is especially difficult in the unmanaged forest area because of the combined effects of geology, soils, permafrost dynamics, and peat. Peat exerts strong controls over hydrothermal regimes and water retention and, where it is thick, can obscure underlying geological material that in themselves can determine water retention and flow patterns (Holden 2005; Heinemeyer et al. 2010). Because geological landforms, composition, and soils differ significantly between the Boreal Plains in the west and the Boreal Shield in the east, their hydrological systems need to be modelled differently (Devito et al. 2005). This is especially important in the Boreal Plains where geological materials are compositionally complex (retain water), topography has low relief, and the landscape is a mosaic of peatland and upland areas with different hydrological properties that interact in complex ways (C. Qualizza et al., personal communication, 2012).

Reducing uncertainties about boreal forest C balances will require well-coordinated interdisciplinary research programs, national-scale data sets on forest conditions and forest change obtained from forest inventories and ongoing monitoring programs (including remote sensing), and ecosystem-modelling approaches supported by advanced computing infrastructure to synthesize and integrate the large volume of data that would be generated by such research programs (Canadian Forest Service, Natural Resources Canada 2012).

## 6. Conclusions

We estimate that, since 1990 Canada's managed boreal forest has acted as C sink of 28 Tg C year<sup>-1</sup>, removing  $CO_2$  from the atmosphere to replace the 17 Tg of C annually transferred out of the forest in timber-harvesting operations and store an additional 11 Tg of C in biomass, dead wood, litter, and soil C pools (Stinson et al. 2011). A large fraction (~57%) of the C harvested since 1990 remains stored in wood products in use and in solid waste disposal sites in Canada and abroad, replacing C emitted from the decay or burning of wood harvested prior to 1990 and contributing to net increases in HWP and landfill C storage pools. The use of these HWP products has contributed to reduced emissions in other sectors where HWP have replaced more emission-intensive products such as concrete, steel, and plastics; but the magnitude of the substitution benefits in Canada and abroad is subject to ongoing research.

Carbon balance estimates for the unmanaged boreal forest are currently limited to "poorly constrained" process models with high uncertainties owing to the lack of forest inventory data and limited understanding of the extent and impacts of permafrost thawing and climate change. In the unmanaged forest, fire is the predominant disturbance type, with very minor insect disturbances, no forest harvesting, and very small areas affected by land-use change. Thus, the unmanaged forest is likely to have been a sink in the second half of the 20th century (owing to low fire disturbances; Kurz and Apps 1999) and has recently transitioned to a smaller sink or a small source as the area annually burned has increased, but this conclusion is highly uncertain.

Biomass C stocks (an indicator that can be used as a proxy for growing stock volume) are increasing slightly in the managed boreal forest, suggesting that current rates of harvesting, natural disturbances, and forest management (e.g., fire suppression, planting, and other silvicultural activities) are sustainable with regard to biomass and total ecosystem C stocks. However, there are regional differences in harvest and disturbance rates, and there may be regions within the boreal zone where the combined impact of human and natural disturbances is currently larger

than the ability of these forests to sustain biomass stocks, in particular in areas with significant impacts on the forest from other industrial sectors. Moreover, sustainability of human actions is not assessed by C stock changes alone. Our conclusions are affected by numerous uncertainties, including uncertainties about rates of regeneration following disturbances and rates of growth, with possible errors in either over- or under-estimating rates of C stock changes.

The single largest threat to C stocks and future C balances in Canada's boreal forest is human-caused global climate change. Large C stocks have accumulated in the boreal zone because decomposition of organic matter is limited by cold temperatures and often anoxic environments. Increases in temperatures and disturbance rates could result in a large net C source during the remainder of this century and beyond. Uncertainties about the response to global change factors remain high, but we emphasize the asymmetry of risk that sustained large-scale increases in productivity and C uptake are unlikely to be of sufficient magnitude to offset higher C losses from increases in area burned and heterotrophic respiration (Kurz et al. 1995, 2008a; Metsaranta et al. 2011).

Reducing the uncertainties of the current and future C balance of Canada's 270 Mha of boreal forest requires addressing gaps in monitoring, observation, and quantification of forest C dynamics, with particular attention to Canada's 125 Mha unmanaged boreal forest with large areas of deep organic soils, peatlands, and permafrost containing large quantities of C that are vulnerable to the impacts of climate change.

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## Appendix A: Land-use change data sources

### Derivation of total area of land-use change in the boreal zone

Data used to estimate the reported 11.8 Mha of land conversion throughout Canada's boreal forest during 1990–2008 (see section 3.2.3.1) include a 1990 land-use map (Hill et al. 2011) derived from a rule-based conversion of several data sources (e.g., National Topographic Database and Earth Observation for Sustainable Development of Forests (Wulder et al. 2008) and Geocover (Koeln et al. 2000) land cover maps), a ca. 2010 crop map produced by Agriculture and Agri-Food Canada (T. Fiset, personal communication, 2011), and information on the land

area flooded by hydroelectric reservoirs (e.g., Eichel and Leckie 2006; Lee et al. 2011) (see section 3.2.3.1).

### Details of annual deforestation 1990–2008

Table A1 summarizes the estimated deforestation rate by industrial category from 1990 to 2008.

### Methodology used for deforestation estimates

Deforestation estimates are derived from a national monitoring system operated by the Canadian Forest Service that generates national deforestation estimates for each year from 1970 to present (Leckie et al. 2006, 2009). It is based on interpretation of Landsat satellite images for three time periods (ca. 1975–1990, 1990–2000, and 2000–2008). Additional information is used such as historical aerial photographs; recent high-resolution satellite images; geospatial information such as forest inventory, hydroelectric development, and oil and gas infrastructure data; and sometimes verification through aerial observation or ground visits. Where forest cover loss is related to direct human-induced land clearing for nonforest land use, the area is recorded as deforested, predisturbance forest cover identified, and the sector or industrial category responsible for land clearing recorded. Mapping is conducted on a network of sample cells with a sampling intensity of 4%–12% of the area in the southern boreal forest. In the northern boreal forest where deforestation is extremely rare, mapping is conducted only around known high-activity areas and large events such as mines, hydroelectric developments, and transmission lines, which are easily detected on satellite images. The mapping in each time period is scaled according to sample intensity and interpolated over the three time periods to give annual deforestation estimates.

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Table A1 appears on the following page.



**Table A1.** Area (kha) in Canada's boreal zone annually deforested by industrial class responsible for deforestation (1990–2008).

Industrial category	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	Average per year	Sum
Agriculture	29.1	26.3	23.4	20.6	17.7	14.8	14.4	14.1	13.7	13.3	12.9	12.5	12.1	11.7	11.4	11.4	11.4	11.4	11.4	15.4	293.5
Hydro flooding																					—
Flooded standing forest	—	—	—	34.2	0.7	—	—	—	—	—	—	—	8.7	—	—	0.5	27.6	—	—	3.8	71.7
Hydro reservoir	0.9	1.6	2.1	0.6	—	0.0	—	0.4	3.1	5.4	—	—	—	2.1	2.7	4.5	—	—	—	1.2	23.4
Forestry	2.4	2.4	2.5	2.5	2.5	2.6	2.7	2.7	2.8	2.9	3.0	3.1	3.1	3.2	3.3	3.3	3.3	3.3	3.3	2.9	54.9
Municipal	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.7	0.7	0.7	0.7	0.7	0.8	14.8
Oil and gas	5.3	5.4	5.5	5.6	5.7	5.8	6.3	6.9	7.4	7.9	8.4	9.0	9.4	10.0	10.5	10.5	10.5	10.5	10.5	7.9	150.8
Other																				—	
Hydro infrastructure	1.7	2.2	2.3	1.8	1.0	1.0	1.3	1.3	1.1	0.9	0.5	0.5	0.6	0.8	1.0	0.8	1.3	1.5	1.3	1.2	23.1
Industry	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	4.5
Mining	0.8	0.9	0.9	0.9	1.1	1.1	1.3	1.1	1.1	1.1	1.1	1.2	1.1	1.1	1.1	1.0	1.0	1.0	0.9	1.0	19.9
Peat mining	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.8
Recreation	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.2	2.9
Transportation	0.9	0.9	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.7	0.7	0.7	0.6	0.6	0.6	0.7	0.7	0.6	0.7	14.1
<b>Total</b>	<b>42.5</b>	<b>40.9</b>	<b>38.9</b>	<b>68.3</b>	<b>30.8</b>	<b>27.6</b>	<b>28.2</b>	<b>28.5</b>	<b>31.2</b>	<b>33.5</b>	<b>27.8</b>	<b>28.1</b>	<b>37.0</b>	<b>30.6</b>	<b>31.6</b>	<b>33.6</b>	<b>56.8</b>	<b>29.3</b>	<b>29.0</b>	<b>35.5</b>	<b>674.2</b>